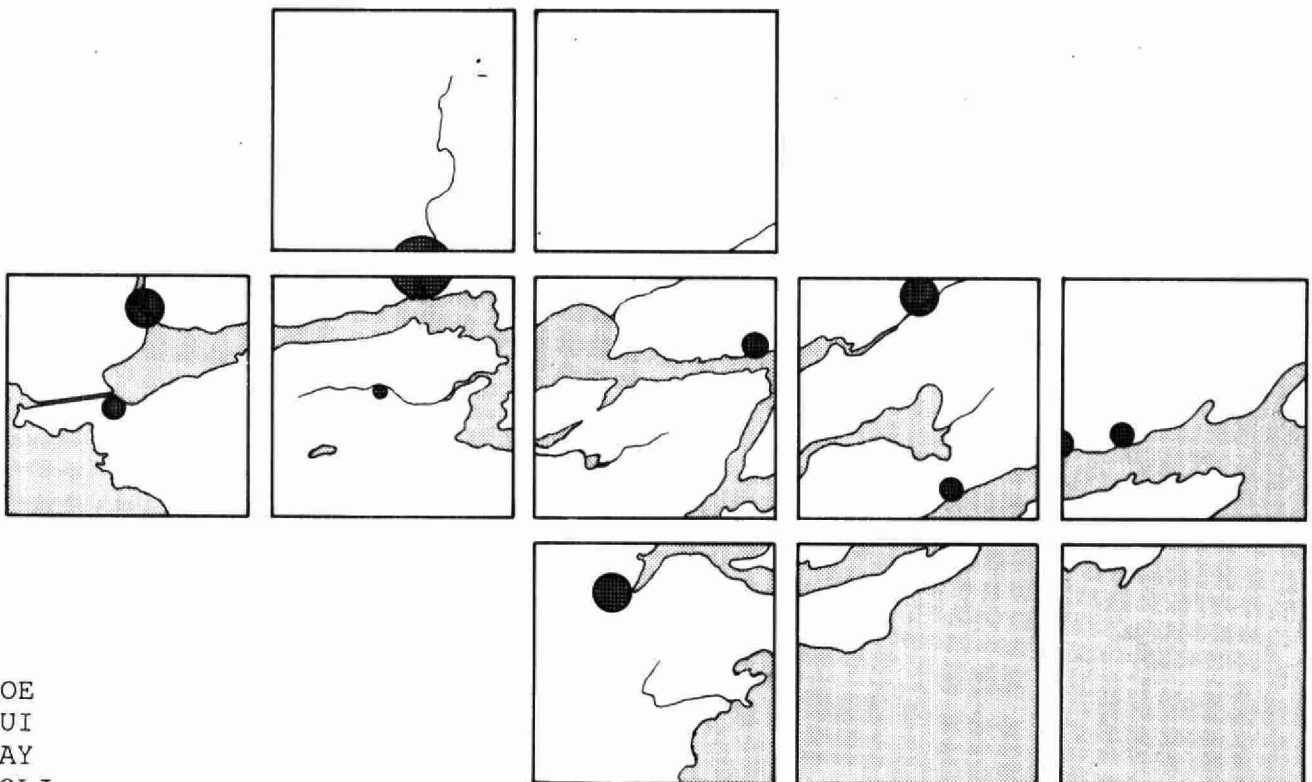# Bay of Quinte Remedial Action Plan

Feasibility of

Decreasing the Internal

Phosphorus Loadings

in the

Upper Bay of Quinte

Technical Report No. 7

BAY OF QUINTE REMEDIAL ACTION PLAN

FEASIBILITY OF

DECREASING THE INTERNAL PHOSPHORUS LOADINGS

IN THE UPPER BAY OF QUINTE

Prepared For:

The Bay of Quinte RAP Coordinating Committee

Prepared By:

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June 1989

FOREWORD

In its 1985 report to the International Joint Commission (IJC), the Great Lakes Water Quality Board recommended that the appropriate jurisdictions prepare and submit detailed Remedial Action Plans (RAPs) for the restoration of beneficial uses of 42 identified 'Areas of Concern' on the Great Lakes System. The Bay of Quinte is one of the IJC identified 'Areas of Concern'.

The process of developing a RAP for the Bay of Quinte was initiated in 1986 with the formation of a Federal/Provincial Coordinating Committee to oversee preparation of the RAP.

The Coordinating Committee in its February, 1987, Progress Report defined excessive nutrient enrichment, persistent toxics and bacteriological contamination as the factors responsible for the impairment of Bay of Quinte beneficial uses. It also identified technical data gaps and a list of potential options which required study.

This report is one of a series of follow-up technical reports. The feedback of phosphorus from the bay sediments is a significant factor contributing to the excessive level of nutrient enrichment of the bay waters. This report provides an evaluation of the feasibility and cost of decreasing the feedback of phosphorus from the bay sediments.

It should be noted that this report is only intended to serve as a background reference document. It provides useful information that will assist the Coordinating Committee and the public in evaluating the options and, ultimately, defining a remedial action plan for the bay.

This option should not be screened in isolation; rather, all the options being considered at this time should be integrated and, thus, analyzed in an ecosystem context.

The Bay of Quinte Remedial Action Plan, when completed, will contain a summary of this report outlining the general findings of the report.

June, 1989
Bay of Quinte RAP Coordinating Committee

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EXECUTIVE SUMMARY

1.   Study Scope

In this study, options for controlling internal loading have been considered. The options considered are:

o   the Null Option (Natural Clean-up)
o   Chemical   -   Aluminum Salts
                    -   Ferric Salts
                    -   Calcium Salts
o   Biochemical Control
o   Fly Ash
o   Barriers
o   Dredging
o   Sediment Burial
o   Aeration
o   Lake Drawdown

The control of internal loading could potentially have a substantive impact upon the state of eutrophication of the Upper Bay of Quinte because the flux of phosphorus from the sediments is large compared to that from land-based sources. This relative effect is illustrated by the magnitude of the fluxes.

Flux	Summer Period	Whole Year
External Load	1.2 g $P/m^2/d$	2.3 g $P/m^2/d$
Settling	2.4	2.3
Return Flux	2.6	2.3

Furthermore, the residence time of phosphorus in the Upper Bay is approximately one-half month compared to that of water which is one month.

4111.1   1.

2.   Description of Control Options

The null option involves allowing the Bay to naturally control internal loading. If diffuse and point sources are controlled, fresh sediment containing lower concentrations of phosphorus will bury older sediment and decrease the rate of internal loading. This coupled with sediment erosion and washout will decrease the phosphorus of surficial layers and decrease internal loading over the long term. On the other hand, the benthic communities and macrophytes which take up phosphorus from deep sediments layers and deposit it as fecal material at the sediment-water interface would help to maintain the current level of internal loading for a period of time.

The chemical and biochemical control options involve addition of various salts to achieve control. Aluminum and ferric salts are injected into the lake water; they form hydroxide precipitates which inactivate phosphorus in the sediments. Calcium salts such as calcite sorb phosphorus and retain it in the sediments; it may also form a calcium phosphate precipitate to activate the phosphorus.

Biochemical control has similar effects to that of aeration - the thickness of the oxidized sediment layer is increased by both methods. Biochemical control depends upon nitrate diffusing into the sediments to increase the oxidized zone thickness while aeration increases the oxygen contained in the overlying water.

The three techniques: fly ash, dredging and sediment burial are all achieved by movement of large quantities of material. Dredging removes the source of the internal loading. Sediment burial decreases internal loading by capping the sediment with clean soils or sediment and increasing the diffusion distance for phosphorus. Fly Ash is a particular material used for sediment burial; it also provides additional protection due to its large sorption capacity for phosphorus.

Barriers are curtains, booms, and/or macrophyte beds placed in the lake water column to decrease current velocities. Curtains could have an impact, if sediment erosion is a substantial cause of internal loading.

3.   Report Organization

In this report, the options are considered as follows. The focus of this study and the attributes of the Bay of Quinte are considered in Section 1. Each option is described in Section 2. The logistics of implementation of each option are considered in Section 3 and the costs in Section 4. Then the detrimental impact of each option is described in Section 5. The potential effectiveness of two options (the null option and alum injection) are considered in Section 6 using a modelling and experimental approach.

4.   Assessment of Effectiveness

The effectiveness of only two control options: the null option and alum injection was assessed in this study. The main approach used for assessing the null option was a modelling approach. There is an excellent data base available for model construction. However, substantial imagination and insights from other modelling studies were used in developing the final model.

The assessment of the effectiveness of the null option suggests that the null system is a legitimate control option with the sediment response expected within a decade or less of land-based phosphorus control. The effectiveness of the null option is, thus, solely a function of the effectiveness of control of land-based impacts. The effectiveness of aluminum injection may be a transient one, only decreasing a portion of the internal loading. Two modelling estimates place this degree of control achieved by alum at a reasonable level to a substantial level. These two ranges result from large uncertainties in the absolute magnitude of internal loadings, and the impact of aluminum as a coagulant upon sediment erosion.

5.   Comparison on Options

These options are compared in Tables 1 to 3 with respect to their relative ease of implementation (Table 1), the cost of implementation (Table 2), and detrimental effects upon the social and natural environment (Table 3). The costing data, in particular, must be treated as order of magnitude because it is based upon literature data which describe small scale operations (50-100 ha). Greater economics should be anticipated for application to the larger scale Upper Bay of Quinte (13,000 ha).

TABLE 1:	RELATIVE EASE OF IMPLEMENTATION	

Option	Implementation	Relative Ease of Implementation
1 Null Option	No logistics	1
2 Chemical A Alum B Ferric C Calcium	Application every 3 years by a flotilla of small boats in liquid form Application yearly by a similar flotilla	5
3 Biochemical	Application annually from a flotilla of small boata in granular form	5
4 Fly Ash	Application with an unknown frequency. It requires substantial quantities of material	8
5 Barriers	Placement and Removal of Barriers every year	7
6 Dredging	Removal of sediments with a smaller number of barges (relative to alum application)	2
7 Burial	Similar to dredging	2
8 Aeration	Placement and maintenance of aeration equipment. Less labour intensive than most other methods, but circulation is quite power intensive	4
9 Lake Drawdown	Cannot be implemented	-

TABLE 2: COST OF IMPLEMENTATION

Option	Annual Operating Cost	Frequency of Operation	Comments
Null Option	0	0	No direct costs
Chemical (A) Alum (B) Ferric (C) Calcium	8 million 5.6 million 7 million	1/3 years 1/3 years 1/year	Major difference of alum and ferric are higher per unit costs for ferric but higher dosage rates for alum Substantially cheaper than Al or Fe.
Biochemical	100 million	1/year	Frequent application
Fly Ash	107 million 25 million	1/year 1/10 years	Appropriate application rate needs to be calculated based upon sediment modelling and results of point/diffuse source content
Barriers	2.2 million	1/year	Placed and removed each year. Assumes application to only one-half of the Bay. Effect on internal loading is not considered.
Dredging	28 million	1/10 years	Frequency of dredging needs to be more fully evaluated.
Burial	4 million	1/10 years	Detailed analysis; 50% safety factor gives $6 million/year.
	12 million	1/10 years	Based on typical dredging costs for Hamilton Harbour.
Aeration (a) Enhanced Circulation	75 million/yr	every year	Cost differences are caused by number of installations assumed; enhanced circulation assumes 272 compressors.
(b) Oxygenation	1.1 million/yr	every year	Oxygenation assumes 10 direct injection units.
Lake Drawdown	Not costed	-	Infeasible.

*Capital cost components amortized at rate of 12% per annum.

TABLE 3:   COMPARISON OF DETRIMENTAL EFFECTS

Control Option	Detrimental Effects
1 Null Option	o Continued loading maintains stressed ecosystem.
2 Chemical A Alum	o Generally negligible except for potential toxicity of aluminum
B Ferric	o Negligible
C Calcium	o Negligible
3 Biochemical	o Nitrate addition may affect ecosystem
4 Fly Ash	o Smothers existing benthic community o Heavy metal content is potentially toxic o Turbidity during placement o Altered sediment transport patterns of fine-grained fly ash
5 Barriers	o Potential impact upon boating o Social concerns o Movement of bottom anchors will disturb sediments
6 Dredging	o Disturbs benthic community and fish habitat. o Potential impact of suspended solids and release of trace metals and toxic organics; field data show that these concerns can be minimized with well designed operations and disposal areas.
7 Sediment Burial	o Effects similar to fly ash (except no heavy metal concerns) and dredging
8 Aeration	o Minimal effect, except it has the potential to disturb sediments if not properly installed
9 Lake Drawdown	o Not evaluated

For some techniques, the costs are obviously so excessive that even their consideration in this study could be construed as incredible for the RAP. This was not the objective of this study and so the reader is accordingly forewarned. The approach of the study was that of environmental systems analysis: to consider all alternatives in a screening fashion irrespective of excessive potential costs, potential detrimental effects upon the environment, or potential logistical problems.

In the Northern American and European field of Lake Management, two options, dredging - historically and presently - alum injection, have enjoyed the widest application. However, many of the others (chemical treatment, biochemical control, fly ash, aeration, lake drawdown, and flushing) have been used. The literature seems to indicate that a higher benefit-cost ratio is achieved with aluminum injection in comparison to dredging. Dredging has historically been used because it removes the source of contamination. Furthermore, a substantial amount of work has shown that the environmental impact of dredging is much smaller than historically feared and has demonstrated techniques for minimizing the impact of both dredging and disposal operations. The high benefit-cost ratio for alum has recently been supported by the effectiveness of alum treatment in two lakes which were monitored for 10-12 year periods. The one potential detriment for aluminum documented in this report is the potential toxicity of Al ion. However, the residual levels should be below regulatory limits in lakes with a pH range of 6-9 and would be similar to levels in water treatment plants using alum coagulation. Further specifics on detrimental effects can be obtained by reading Section 5.2a (Detrimental Impacts of Alum Injection) and Section 5.6 (Detrimental Impacts of Dredging).

6. Further Studies

Further studies are required to increase our understanding of internal loading and to delineate the implications of alternative remedial and mitigating measures being considered in the Remedial Action Plan. Further studies should centre upon upgrading the evaluation of the effectiveness of the null option option given in this report and of a control option such as alum injection. Specific studies are indicated in Section 6. They include:

o   laboratory work

- studies to address deficiencies in the alum dosages used in our study and its effects upon pore water phosphorus concentration. Specifically, laboratory testing of alum over a wider range of dosage to:
  . confirm the data of this report, and
  . evaluate competition for phosphorus by cations such as calcium; and
- studies to measure the sediment resistance (sediment shear strength) to erosive action using readily available techniques.

o   field work

- further field measurements in the Upper Bay of Quinte using the technique of Sly to confirm the magnitude of the internal cycling of phosphorus; and
- further field measurements (peepers, sediment dating, methane dynamics, bioturbation, oligochaete densities) to provide data for checking the model's evaluation of natural restoration;

o   modelling studies

- further chemical equilibrium and mineralogical studies in support of the modelling assessment;
- model revisions to include other processes and observations to further validate the geochemical based model for phosphorus return;

o   pilot scale trial of alum injection

- pilot-scale testing of alum to confirm the several deficiencies which result from a lab scale/modelling activity.

Pilot scale testing after further laboratory testing and modelling would be the appropriate approach. Only pilot scale will confirm the results from the laboratory. Pilot scale work would be undertaken if the laboratory results are favourable for showing that coagulation increases the sediment resistance to erosive action.

7.   Implications for the Remedial Action Plan

This report can only indicate a few of the implications.

This study team has reviewed the overview synthesis report, the various component reports and had conversations with various members of the RAP team. However, this study team can only indicate a few of the implications of the results of this report in a knowledgeable way. They include the following:

i)   There are still considerable uncertainties in the magnitude of internal loading in the Bay of Quinte. Further studies outlined in this report are required to confirm the probable magnitude.

ii)   The null option is a legitimate control option which, if coupled with improved land-based control of phosphorus, should show a decrease in internal loading in the Bay of Quinte.

iii)   The degree of decrease of internal loading would not appear to be large given the probable magnitude of reduction of inputs associated with further point source control. An additional decrease would result from control of diffuse sources. A modelling exercise using the tools developed in this report and incorporating aspects of the overview Ecosystem Phosphorus Model, would increase the confidence of the RAP team in setting of priorities for control of internal loading.

iv)   Of all the other control options considered (chemical/biochemical, aeration, fly ash, dredging, sediment burial, barriers, and macrophyte beds), these study results suggest that a chemical technique such as alum injection would be the next priority as a control option for controlling internal loading.

v)   Due to a number of factors and data deficiencies, alum injection would not appear to be a feasible option in the short term but should be considered for the long-term. To aid the assessment of its long-term possibilities, a limited number of further studies, consisting of lab-scale studies, field measurements, modelling studies, and a pilot scale-trial application, should be conducted.

vi)   A few control methods evaluated in this study, may be locally quite effective to remediate internal P loadings and other contaminant concerns. Their effectiveness for such applications requires evaluation, before implementation.

1.0   BACKGROUND AND OBJECTIVES

1.1   Introduction

Internal loading of phosphorus to lake water from the sediments may represent a significant source of phosphorus to algal populations during the summer. This has been demonstrated in the Upper Bay of Quinte where internal loading of phosphorus has been calculated to exceed external inputs of phosphorus (Minns et al., 1986). This quantification is approximate. To estimate internal loading more precisely, BEAK is presently undertaking a measurement program to characterize and refine the return fluxes of phosphorus. This will provide key insights into the mechanisms of return and the feasibility and effects of various control options.

As a part of a Remedial Action Plan, the Coordinating Committee commissioned this study for "determining the feasibility of decreasing the severity of the sediment phosphorus release phenomenon".

1.2   Objectives

This study component has two objectives:

1.   to determine the feasibility of various control measures to minimize sediment phosphorus release (e.g., chemical or physical stabilization); and

2.   to determine the costs over the long-term (several years) as well as an analysis of possible detrimental effects (i.e., effects on bottom fauna and fish).

1.3 Mechanism Controlling Internal Loading

Several mechanisms have been proposed to explain the release of phosphorus from lake sediments:

1. diffusion of dissolved interstitial or desorbed sedimentary phosphorus into the overlying water, primarily under anaerobic conditions;

2. aerobic decomposition of freshly sedimented material, including algae and macrophytic debris, at the sediment-water interface; and

3. erosion of sediment phosphorus by lake currents.

Phosphorus release from sediments is affected greatly by redox conditions at the sediment-water interface and by fluid velocity in the water near the sediment-water interface. Early studies by Mortimer (1941, 1942) in several British Lakes showed that, during periods of anoxia in bottom waters, phosphorus was desorbed from particulate sediment phases and released into the water column. This process was related to the reduction of iron and subsequent breakdown of ferric-phosphate complexes. These and subsequent studies solidified the belief that phosphorus release occurs only under anoxic conditions. This concept was reflected in mathematical models of phosphorus dynamics appearing in the 1970's and early 1980's where the sediments were envisioned as a sink for phosphorus from which the rates of return to the water column were negligible under generally aerobic conditions (Dillon and Rigler, 1974; Dillon, 1975; Snodgrass and O'Melia, 1975; Vollenweider, 1969, 1975; Lund et al., 1976; DiToro and Connolly, 1980; Snodgrass and Dillon, 1983). Under anoxic periods during summer or winter stratification, phosphorus return was foreseen as substantial.

However, further investigations have shown that phosphorus release does indeed occur under oxidized conditions, although at a much lower rate than under anaerobic conditions. Table 1.1 summarizes the results of relevant studies where phosphorus release has been measured from lake sediments. Values range from 0.4 to 12 mg/m^2/day and 3 to 31 mg/m^2/day under aerobic and anaerobic conditions, respectively.

Current regimes at the sediment surface also affect the flux of nutrients and chemicals, including phosphorus, across the sediment-water interface. Oxygen uptake in core or in situ incubations is known to increase significantly with stirring speed or water velocity (Carey, 1967; Edwards and Rolley, 1968; Boynton et al., 1981). Silicate release in cores was increased 40 to 50% on stirring the supernatant water (Andrews, 1980). Water currents increase the flux of dissolved materials across the sediment surface by

TABLE 1.1:   EXAMPLES OF PHOSPHORUS RELEASE RATES OBSERVED IN LITERATURE

Lake	Investigator	Oxid Rate $(mg/m^2/d)$	Anoxic Rate $(mg/m^2/d)$	Comments
Erie Central B	Burns and Rom (1970)	0.68	7.4	Phosphorus Budget Calculations ($10\text{-}12^oC$)
Lough Nugh Scotland	Rippey (1976)	2	-	o 20^oC; DO = 100% saturation. o Release was pH dependent. o Lab Experiments on Intact Cores
Castle	Neame (1976)	0.65	-	Box Corer, P-32 Tracer, Sediments from 4 m.
Mendoth Wisconsin	Lee et al. (1976)	-	-	o Oxic rate was 10% of anaerobic rate. o Attributed oxic return to decomposition of freshly sedimented matter.
Warner Mass.	DiGiuano and Snow (1976)	6	-	o Reservoir, 2 = 1.6 m; tw = 10 d. o In situ measurement, lab experiments and modelling study.
Warner Mass.	Fillos and Swanson (1975)	0.4	8.5 (max.)	o Lab study using mass balances on CSTR over sediment. o Phosphorus release correlated with Fe release under anoxic conditions.
Muddy River Mass.	Fillos and Swanson (1975)	3.1	31 (max.)	Muddy River is same flow system as Warner Lake.
Baldegersee Switzerland	Vollenweider (1968)	-	3 to 4	In situ study.
Poboy Sound	Pomeroy et al. (1965)	0.03	-	Estuary.

TABLE 1.1: EXAMPLES OF PHOSPHORUS RELEASE RATES OBSERVED IN LITERATURE

Lake	Investigator	Oxid Rate $(mg/m^2/d)$	Anoxic Rate $(mg/m^2/d)$	Comments
Esrom Denmark	Kamp-Nielsen (1976)	4.2	-	Modelled by simulation sedimentation, resuspension and exchangeable fraction on sediment.
Esrom Denmark	Kamp-Nielsen et al. (1982)	11-13	-	o Examined effect of bioturbation on return rate. o One species increased the return rate which the other decreased the rate 4%.
Erie	Lam et al. (1983)	-	-	Resuspension Modelling Study. See text for rate.

decreasing the boundary layer thickness (Boudreau, 1981; Boudreau and Guinasso, 1982). Theories are being developed to relate the effects of fluid mechanics on fluxes of phosphorus, oxygen and metals from sediments. Currents of extreme velocity may cause erosion of the sediment surface with the subsequent release of phosphorus from suspended particles (Lam et al., 1983).

A sketch of factors affecting phosphorus release to the water column is given in Figure 1.3. The figure is divided into zones denoting:

o   erosional processes (I),
o   diffusional processes (II),
o   dredging processes (III), and
o   aquatic organisms (IV),

as the major processes affecting phosphorus transport from the sediments to the water column.

In addition, the food chain transfers phosphorus from the sediments to the water column via feeding activities. Benthic organisms will ingest sediment particles containing organics, bacteria, metals and phosphorus; readily adsorbed phosphorus may be accumulated in the cells of these organisms. Feeding on benthic organisms by higher members of the food web (eg. fish), transfers the phosphorus to the higher levels. The importance of this feeding activity relative to other sediment release mechanisms is of potential concern for the development of management protocols. An investigation using food transfer models, literature bioconcentration factors and estimates of aquatic biomass (benthic organism density), members of feeding fish, and of macroinvertebrates could be used to compare the relative impact of organism feeding and macrophytes on the transfer of phosphorus from the sediments to the water column. Such phenomena are beyond the scope of the work of this study.

The bottom currents, wave action and other turbulent processes (section 3, Figure 1.3) erodes sediments transporting particulate and soluble porewater forms of phosphorus into the overlying water (arrow 1, Figure 1.3) (Lick, 1982; Sly, 1982; Kenney, 1985). In the water column, desorption of phosphorus (arrow 2,) under aerobic conditions makes the phosphorus particularly available to aquatic organisms.

FIGURE 1.3:   SKETCH OF MECHANISMS FOR TRANSFER OF PHOSPHORUS
FROM SEDIMENTS TO WATER
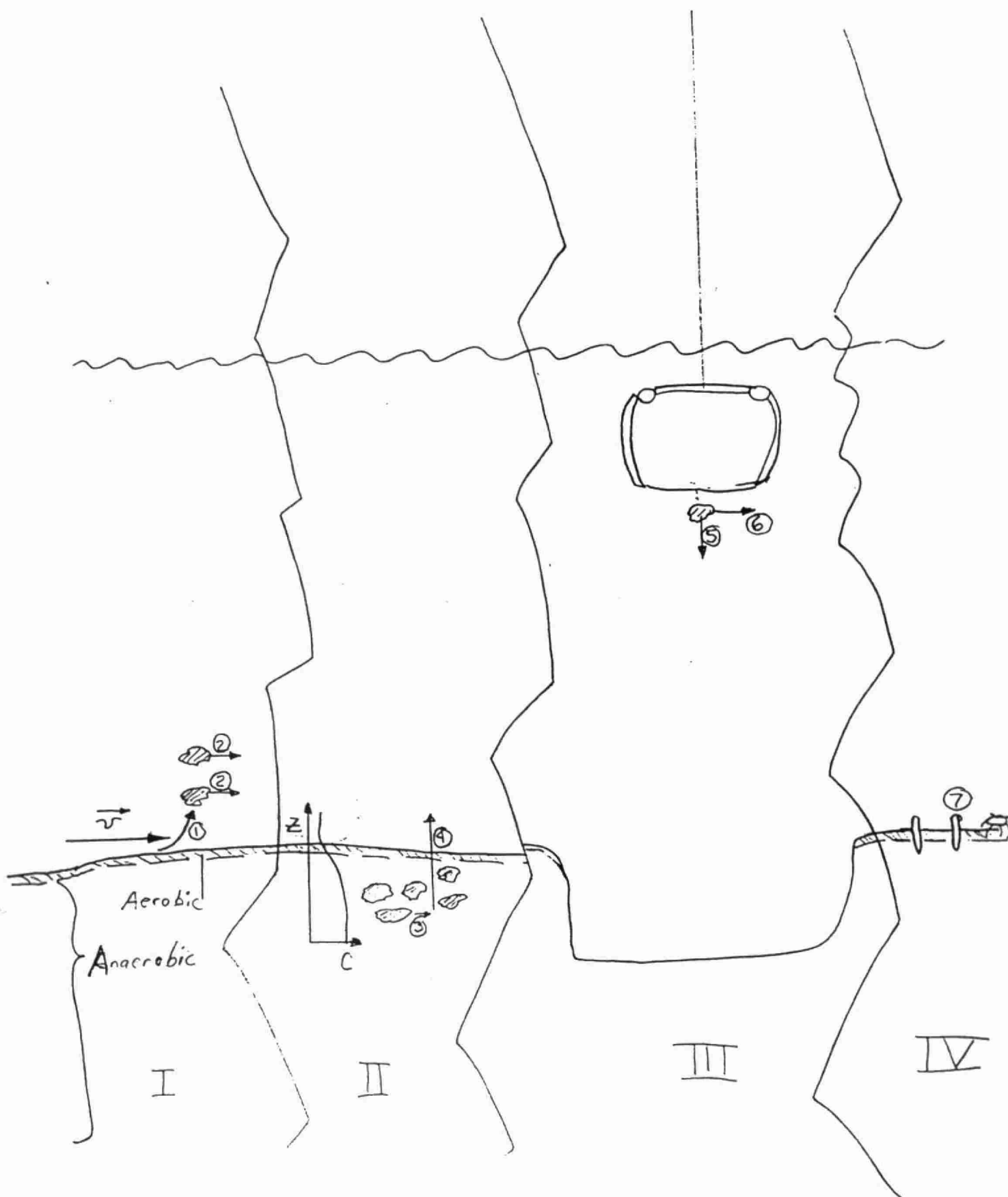
In the sediments, the desorption of phosphorus under predominantly anaerobic (arrow 3) and also aerobic conditions (the surface oxidized zone) introduces phosphorus into the lake water. If a concentration gradient such as depicted in Figure 1.3 (section II) results, the phosphorus will diffuse in the overlying water.

Transport of phosphorus from sediments is also catalysed by benthic organisms reworking the sediments (arrow 7, Figure 1.3). This occurs by organic ingestion from deeper sediments of sediment particles and their defecation into the surface sediments and by an increase in the effective diffusion coefficient (i.e., larger than molecular scales) in surficial sediments (Robbins, 1982; Kamp-Nielson et al., 1982; Snodgrass and Walker, 1985).

Other human activities (dredging, propellors of ships) removes phosphorus from sediments (Sly, 1977). Loss of sediment material during dredging (as depicted by arrow 5 for a clam bucket, section III, Figure 1.3) makes the phosphorus content more readily available to the water column. Subsequent desorption (arrow 6) increases availability to aquatic organisms. Such effects as boat propellers can be viewed as a bottom loss effect (arrow 1), albeit consideration of the fluid mechanics involved is also required. Detailed analysis of such fluid mechanics is beyond the scope of this investigation; mechanical considerations would need to be synthesized for application to the Bay of Quinte. Such considerations are given and have been evaluated by Aalderink et al (1985).

1.4 Control Options

The literature on control options, their feasibility, their effectiveness, their costs and detrimental effects has increased substantially in recent years (e.g. Peterson et al., 1973; Dunst et al., 1974; Garrison and Knauer, 1983). The feasible options which are considered generally include:

- o   null option (allow the lake to clean itself up after reduction of external P loading);
- o   chemical precipitation (calcium, aluminum, iron precipitation of phosphorus in the sediments);
- o   biochemical control (increase depth of aerobic sediment to decrease phosphorus flux);

o   fly ash (adsorption of phosphorus);

o   barriers to current induced erosion (to prevent erosion of sediments);

o   dredging (removal of sediments with excessive phosphorus concentrations);

o   sediment burial (addition of clear fill);

o   reaeration (to maintain high oxygen levels in the lake and hence prevent anoxic conditions which increase the internal loading); and

o   lake drawdown (to maintain aerobic sediments and hence suppress phosphorus release).

For the Bay of Quinte, these options for the RAP can be broken down into three categories:

1.   Options Which Obviously Should Be Evaluated

2.   Options Which Are Less Plausible

3.   Options Which Are Infeasible (which can arbitrarily be eliminated)

Detailed evaluation given below allows decisions to be made about the feasibility of applying these techniques.

1.5   Characteristics of the Bay of Quinte

The characteristics of the Bay of Quinte pertinent to this study are morphometric characteristics, land use factors which affect control options or which would be detrimentally impacted by particular control options, and chemical characterization of the sediments. Chemical characteristics include whether internal loading is constant Bay wide and whether the elevated phosphorus levels are Bay-wide, or only in local areas.

The Bay of Quinte and its watershed are given in Figure 1.5.1. The Upper Bay of Quinte, which is the subject of this report, is generally located between Trenton and Deseronto (see Figure 1.5.2). It is shallow (mean depth 3.5 m; maximum depth 14.9 m; general depth of 4-8 m), and generally well mixed due to surface wind stresses. It has a substantial surface area (13,600 ha) but a relatively small water volume (476.5 million m^3) due to its shallow mean depth. It is long and narrow, being of the order of 35 km in length and varying from 1 km to 4 km in width. Its average annual detention time is estimated to be 1 month.

Legend:
- Trent
- Moira
- Salmon
- Napanee
- Prince Edward County

MARMORA Fe (1840-1890)
GUNTER Pb
DELORO Fe, Ag, As, Au (1907), Co (1914-1961)
ELDORADO Au (1866), Cu (1877)
MADOC Fluorspar (1912) Talc (1899) Fe (1837-1910), Pb (1898-1906)
TWEED Limestone, Marble Granite, Slate Cement (1890)

SULPHIDE 8 Sulphide Mines
MARLEBANK Marl (1900)

CAMDEN EAST Paper Mill
NEWBURGH Tannery (1832), Paper Mill (1880)
STRATHCONA Paper Mill (1870)

MARMORA Bark Mill, Tannery (1821)
FRANKFORD Tannery and Mill (1820)

TRENTON Iron Works (1875), Pulp Mill (1880), Munitions (1916-1918), Creosote (1917), Newsprint (1927), Pulp Mill (1956)
CORBYVILLE Distillery, Molasses Sludge
BELLEVILLE Chemicals (1949)

BATH Cement (1978)
STRATHCONA Cement (ceased 1904)
PICTON Cement (1958)
POINT ANNE Cement (1905-1970)

Pigeon L.
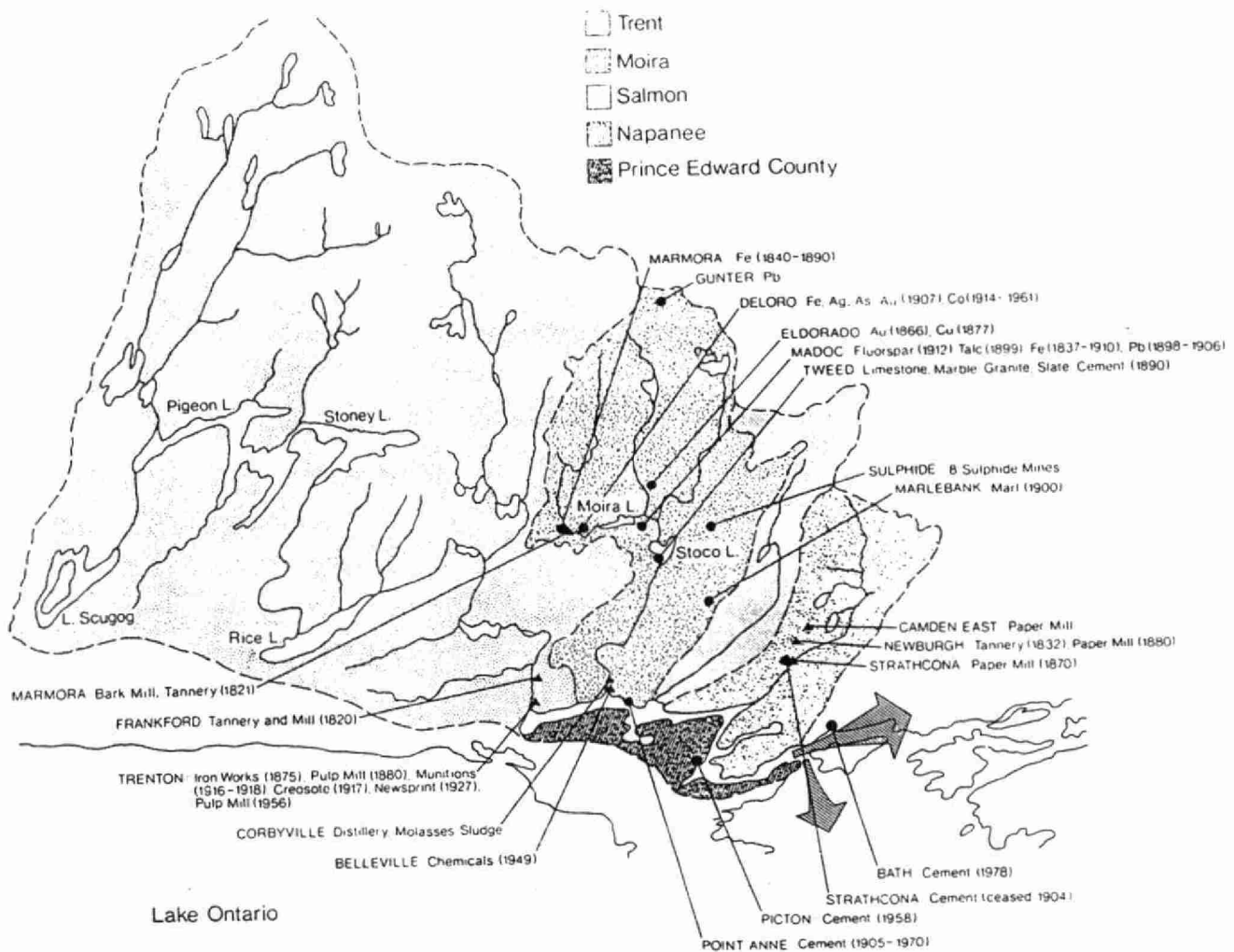Stoney L.
Moira L.
Stoco L.
L. Scugog
Rice L.
Lake Ontario

FIGURE 1.5.1: Minerals, Mining, and Quarrying in the Bay of Quinte Watershed (Modified After Moira River Conservation Report 1950; Napanee River Conservation Report 1957; Prince Edward County Conservation Report 1968; Lower Trent Region Conservation Report 1970)
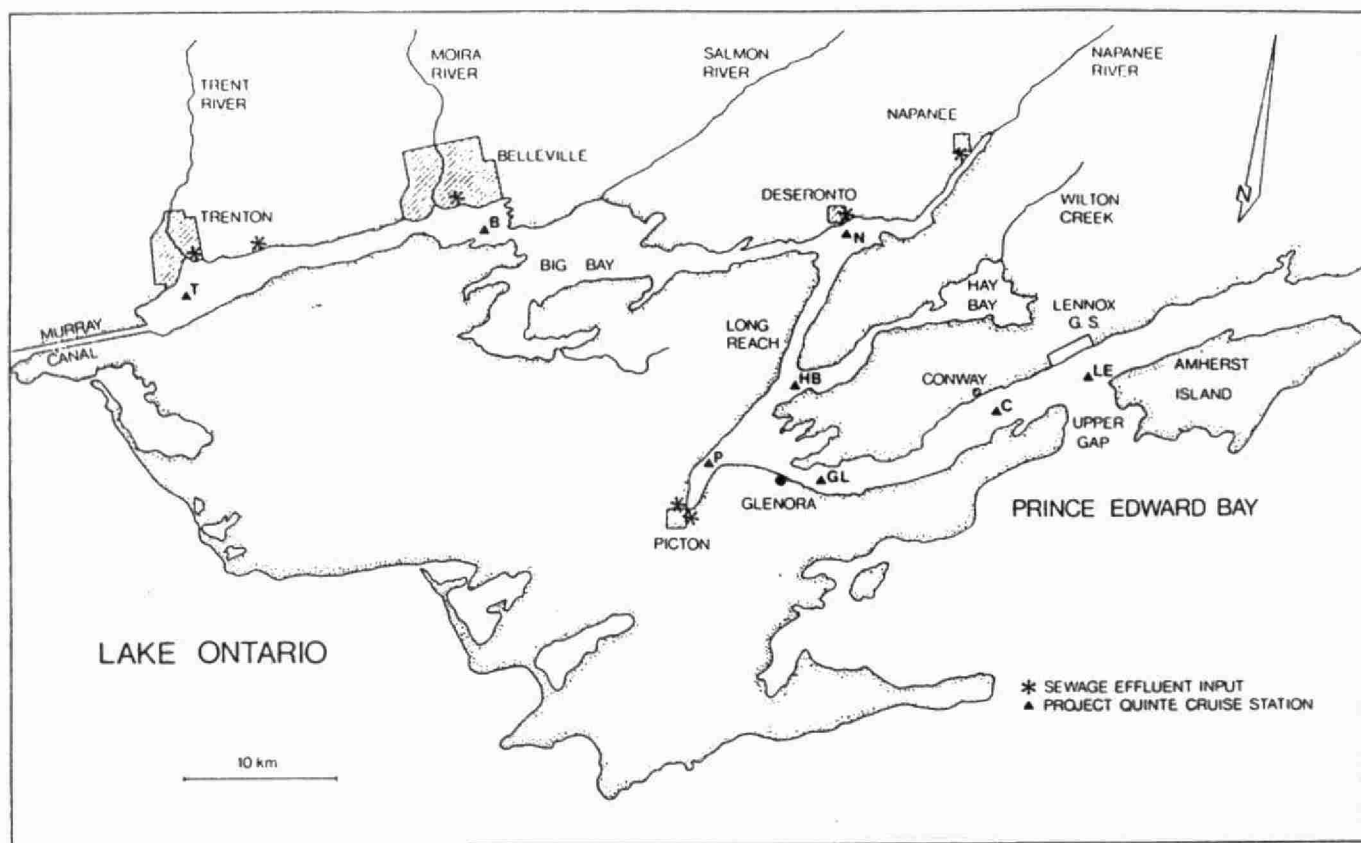
FIGURE 1.5.2:   Location of the Main Sampling Stations in the Bay of Quinte

In comparison, the middle and lower Bay are subjected to varying degrees of stratification and also to hydrodynamic exchange with Lake Ontario. This dilutes the elevated levels of phosphorus and increases the oxygen content of the hypolimnetic waters. The middle and lower portion of the Bay are also deeper, resulting in a mean depth for the whole Bay of 10.4 m (1972-1981), a surface area of 257 km^2, a volume of 2.67 km^3, a net length of 64 km, and a drainage area of 12,600 km^2 (hence DA/SA = 49).

The state of eutrophication of the Bay is a prime concern of the Remedial Action Plan. Under the assumption that phosphorus controls the state of eutrophication over the long term (which has been demonstrated quite ably by Lean and others) control strategies are focussing upon phosphorus dynamics and the resultant ecosystem changes. Of particular concern have been the excessive phosphorus concentrations (typically 70-80 ug/L) which resulted from an areal phosphorus loading of approximately 2.5 g/m^2/yr. and chlorophyll a concentrations of the order of 30 ug/L in the Upper Bay in the 1970's (prior to 1977). This has lead to focussing on the control of point source, diffuse source and the internal loadings of phosphorus. With the implementation of phosphorus control in Ontario in two forms:

i)   Reduction of the phosphorus content of laundry detergents to 5% (as P_2O_5) by January 1, 1973 which reduced phosphorus inputs entering municipal STP's (and presumably their effluents) by 50%;

ii)   Implementation and achievement of a 1 mg/L total phosphorus (as P) effluent standard for all wastewater treatment plants (for the Bay of Quinte, the achievement of this standard occurred by early 1978);

the Upper Bay levels of phosphorus were reduced to the 45 to 50 ug P/L range and chlorophyll a levels were reduced to the 15-20 ug/L range. These values are still deemed to be excessive in comparison to desirable levels for phosphorus which are typically in the 10 ug/L range for lakes which have oligotrophic-mesotrophic characteristics and in the 20 ug/L range for lakes which have mesotrophic-eutrophic characteristics.

Associated with these hyper-eutrophic conditions have been other characteristics which are symptomatic of a stressed ecosystem. These include the loss of macrophyte beds for reasons which are not completely clear, loss of fish habitat (particularly that maintained previously by the beds), and potentially increased erosion of bottom sediments due to the changes in protection afforded to sediment by the macrophyte beds.

Unique to this ecosystem is the influence of Lake Ontario upon the lower Bay areas. Exchange flows with the Bay dilute water flowing from the upper Bay to reduce its phosphorus and nitrogen content (closer to the mesotrophic-eutrophic range), its chlorophyll a content, and increase the water transparency. However analysis of data for conservative ions suggests that the lake water does not substantially influence the quality of the upper Bay.

To develop a Remedial Action Plan, the Ontario Ministry of Environment, the IJC, MNR, Environment Canada, Fisheries and Oceans and other concerned agencies have mandated a series of studies to assess the feasibility of controlling the state of eutrophication in the Bay. The four studies of most direct concern are:

i)   Costing and assessment of point-source control (e.g. for STP's at Trenton, Belleville, Deseronto, Napanee, Picton and CFB; industrial sources; and water treatment plants);

ii)   Costing and assessment of non-point source control;

iii)   Costing of methods for dilution; and

iv)   Control of internal phosphorus loading (this study)

An allied study is assessing other contaminants.

An understanding of the relative importance of point and non-point source control, and flushing can be gathered from the work of Minns Owen and Johnson (1986). Of concern to the remedial action plan also is whether the excessive nutrient concentrations (45-50 ug P/L) are maintained mainly by these continuing sources of inputs from land-based sources or also by internal loading. The relative potential impact of internal loading can be assessed by comparing nutrient loading from land based sources to the sediment-water exchange flux. The relative fluxes estimated by Minns Owen and Johnson (1986) are:

TABLE 1.5.1   TP CONCENTRATIONS @ 0-3 CM IN SEDIMENTS IN
MAY JUNE 1966 EKMAN SAMPLES. M.G. JOHNSON

Area	Sample 𝑛	P%	
		Mean Conc	SE
Trenton to Causeway	27	0.41	0.016
Belleville thru Big Bay	56	0.46	0.016
Below Napanee	18	0.44	0.039
Picton Bay area	24	0.44	0.032
Adolphus Reach	17	0.48	0.050

Method was by Technicon Autoanalyser. Check with Ken N. re digestion.

Variety of sediments examined, if Ekman could retain them.

Related to texture, but this is not available on all samples. Most sediments were sandy loams e.g. 56% sands, 31% silt, 13% clay throughout bay. Some shallower samples were 100% sand and fine sand and P would be about 0.1-0.15 in them. However, there were not that many sand samples.

This is all I could do for a quick response.

	For the Summer	On an Annual Basis
Land-Based Loading	1.2 g/m^2/yr	2.3 g/m^2/yr
Settling	2.4	2.3
Internal Load	2.6	2.3

These values suggest that internal loading maintains the observed concentrations during the summer, exceeding the rate of land-based loading. Compared to the hydraulic detention time of 1 month, the phosphorus residence time is approximately 15 days indicating that a net accumulation is occurring in the Bay. Hence a combination of land-based loading control and internal control are necessary to achieve relatively quick results. If internal loading were not immediately controlled, it is plausible that the Bay would still respond to point source control and naturally reduce the internal loading by sediment burial processes. This control option is assessed below under the topic of the "Null Control Option".

There are several considerations which could also influence the efficiency of assessing controls of internal loading. These include the method of point source control, the method of diffuse-source control, the bioavailability of diffuse source phosphorus, and the role of macrophytes in internal loading. If control methodologies similar to that used for Lake Washington were implemented in the Upper Bay of Quinte at least for point sources in which the effluent from all treatment plants is collected and discharged to the Lower Bay, the land-based loading would be reduced by only 18% in the summer and 10% on an annual basis (the current point source loading is 84 kg/day compared to estimates for total land based loadings of 457 kg/day in the summer and 842 kg P/day on an annual basis). This may have a minimal impact. However, for the period of 1970-1980 when the total load was reduced by 23% from 595 kg/day to 457 kg/day in the summer (and a smaller reduction on an annual basis of 4% from 882 kg P/day to 842 kg P/day) and the point source load had been reduced by 72% from 300 kg/d to 84 kg/d, the average phosphorus content of the Upper Bay was reduced by approximately 40%. The higher relative reduction in the water relative to changes in the loadings from point and non-point sources suggests that bioavailability of P may be a significant question which needs to be considered in the whole metabolism of the Bay and of internal loading.

The role of macrophytes in internal loading is uncertain. Present estimates of internal loading are based upon relatively coarse estimates of settling rates (a few insitu measurements with traps and a few measurements in a Bay of Quinte limnocorral) and a back calculation of the internal loading from the phosphorus budget. These estimates of internal loading would include diffusional-gradient-controlled transport from the sediments, aerobic decomposition of fresh seston deposited on the surface of the sediments, erosion of sediment by waves and bottom currents, and macrophyte uptake of P from the sediments and release to the water column. Estimation of internal loading is being made by Beak in an allied study using laboratory techniques. These will characterize diffusional-gradient and aerobic decomposition related internal loading and possibly erosional effects. They will not characterize macrophyte uptake and release. Inspection of the species of macrophytes can provide some evidence for the potential impact of macrophytes - certain species obtain most of their phosphorus from the water column while others obtain theirs from either the sediments or a mixture of the water column and the sediment.

With respect to strategies for control of P, it is plausible that local areas may be more important to loading than the whole Bay. Available information on sediment total phosphorus concentrations are given in Table 1.5.1. These samples are limited in number and were obtained by Ekman Dredge, and used an analytical technique for sediment phosphorus which has subsequently been replaced by a newer technique. An additional set of samples were gathered in the 1980's but, unfortunately, not analyzed. These data are not adequate to allow conclusions concerning whether or not there are local areas (such as near STP outfalls) with excessive P concentrations which require priority attention with respect to control methods. Measurements of internal loading to date do not permit conclusions concerning significant spatial variablity. In fact, given the statistical confidence from the measurements, it is probable that all values will neither be able to differentiate spatial nor temporal variability - the main objective of the measurements are to confirm the magnitude of the estimates of internal loading made by Minns, Owen & Johnson (1986).

1.6   Study Approach

The approach taken in this study for assessing the feasibility of decreasing internal phosphorus loading in the Bay of Quinte, is a comprehensive evaluation of alternatives and a comparison of their respective costs.   This is similar to the approach of environmental impact assessment or systems analysis in which all plausible alternatives are hypothesized, their feasibility is critiqued, costs, benefits and other detriments are assessed.   Then, an overall analysis of the respective options is made, from which the most feasible option is determined.

In this study, it is apparent that certain of the options are infeasible (such as lake drawdown); these options, are a priori discounted. For other options, which are either potentially quite feasible, or which are plausible, a complete analysis of their attriubtes, their feasibility costs, their costs and their environmental detriment is assessed. In some cases, the cost of the option is extraordinary. Such costs may make the consideration of the option incredible to the ordinary lay public. We had several conversations with the steering committee of the Remedial Action Plan, and have concluded that irrespective of how. incredible the cost of the options may be, a thorough analysis should still be performed in this study.   Hence, the strength of the approach of this study is to be thorough for all options, rather than to, a priori, discount the option because of its feasibility or infeasibility. It is left to developers of the overall remedial action plan to provide a synthesis for all study components (point source loading, non-point source loading, internal loading, and flushing) which is credible and which involves only feasible options.

As well, the potential effectiveness all the control options is not assessed because this was beyond the scope of work of the initial study. Two control options are evaluated as a follow-up to the main study using a modelling and laboratory measurement approach.

In this report, control options are described in section 2, the feasibility (logistics) of implementing the control options are described in section 3, the costs of implementation are given in section 4, and finally the detrimental effects of the control options are critiqued in section 5. The potential effectiveness of the null control option and of chemical treatment are evaluated in Section 6.

2.0   DESCRIPTION OF CONTROL OPTIONS

A qualitative description of each option is given in this section. The following headings are used.

2.1   the Null Option - Natural Clean-up
2.2   Chemical   2.2A   Aluminum Salts
                    2.2B   Ferric Salts
                    2.2C   Calcium Salts
2.3   Biochemical Control
2.4   Fly Ash
2.5   Barriers
2.6   Dredging
2.7   Sediment Burial
2.8   Aeration
2.9   Lake Drawdown

2.1   Null Option - Natural Clean-up

The null option represents existing conditions and any long-term changes in internal loading which will occur naturally. Long-term changes in internal loading may result from such measures as point-source or non-point source control, or increases in external loading due to increases in population in the watershed or in the upstream Trent River watershed. Under the mandate of the Remedial Action Plan, it can be assumed that the net effect upon external loads of phosphorus should be a reduction in the loads.

The reduction of external P loads should allow for the reduction of internal loads of P - the major question concerns the time of recovery and the extent of change of internal load. Estimating these parameters (time, extent) represents the forefront of research at the moment albeit available modelling concepts and field data from the Bay of Quinte and other sites can be used to make such estimates. This estimation is beyond the mandate of this report.

2.2   Description of Chemical Methods for Control Options

The purpose of addition of chemicals to a lake is to lower the concentration of phosphorus in the water column by either precipitating it or by preventing its release from the sediments (hence, by inactivating the phosphorus in the pore water of the sediments). Various salts which have been used include aluminum salts (aluminum sulphate, sodium aluminum) ferric and ferrous salts (ferric chloride, ferrous chloride, ferric aluminum sulphate, pickle liquor), and calcium salts (calcium hydroxide, and calcium carbonate). The salts are added to the lake surface or injected into the epilimnion, metalimnion or the hypolimnion. In the case of the aluminum and ferric salts in particular, a precipitate may form to which phosphorus is absorbed or within which it is coprecipiated. Both the aluminum and ferric salts form a hydroxide floc which will emmesh phosphorus within it or which, depending upon the concentration of aluminum or iron, will precipitate the phosphorus. If inactivation occurs, the objective of the exercise is to add sufficient aluminumm, iron or calcium, to create a barrier in the aerobic layer to the release of phosphorus from anaerobic sediments, or to form sufficient precipitates at the sediment-water interface to precipitate or adsorb phosphorus released in the aerobic layer from fresh seston.

2.2a   Aluminum Salts

The first application of aluminum addition to lakes to control phosphorus release was made in 1970 (Cooke and Kennedy, 1981). The guidelines for dose calculation and application were based on the use of aluminum salts in advanced wastewater treatment for phosphorus removal. This technology has been extended to lake rehabilitation but guidelines for dose calculation and application have generally been lacking until recently.

The two main methods of aluminum addition are in the form of alum (aluminum sulphate pentahydrate), and sodium aluminate. Alum is added in relatively alkaline or hard water lakes where there is sufficient alkalinity to neutralize the acidity associated with aluminum to assure a small pH change in the water (for example, the pH may change from 7.8 to 7.4). In the softer water lakes, alum and sldium aluminate have been used (Connor and Smith, 1986). For lakes with very low alkalinity, the addition of aluminum in the form of sodium aluminate provides an additional buffering mechanism to prevent large pH depressions while providing sufficient aluminum content. Typically, sodium

aluminate and alum are added in a 1:1 or 2:1 ratio depending upon the alkalinity of the lake. Application methods have varied as the technique of alum addition has evolved. At the present time there is a very large diversity of application methods. In fact, alum addition for control of internal cycling of phosphorus has become the most widely used technique being tested at the moment. It typically is used in place of dredging due to the high cost of dredging and other environmental impacts of dredging, and the need to find space for disposing of dredge spoil.

Some of the most comprehensive work has been that carried out by Kennedy and Cook (1982) in which P control has been demonstrated for six years. Some of the earlier development of the technique has been that based on the work of Peterson et al., (1973 and 1974).

One of the typical designs used for application of alum is that given in Figure 2.2.1 (after Kennedy and Cooke 1982). In this design, a shore based storage and mixing facility is connected to an application barge via a distribution pipe. The alum is pumped from the storage facility through the distribution pipe onto the application barge and then a manifold based in the water is used to spread the alum as the barge moves across the lake. The barge must include provision for pumping both alum and lake water together to provide a mix to allow flash mixing on the barge prior to discharge of the alum/lake water mixture into the lake. Several different designs have been used to accomplish the mixing. These are described in more deail by Kennedy and Cooke (1982) and others. Using a grid system, then the full lake can be treated with this equipment.

A second design for an application barge which has been used is that of Connor and Smith (1986). In this design, sufficiently large storage tanks were located on the barge to make it self-sufficient and to preclude the need for it to be connected to a storage facility on shore. The barge, given in Figure 2.2.2 (after Connor and Smith, 1986) is a modified weed harvester that has characteristics of manoeverability and capacity to treat a large sediment area. Major modifications to the harvester included the provision of putting tanks on board, and putting the location port on the end of the spray boom to allow delivery of the alum to a particular depth.

In the literature, generally application has been to small lakes of the order of 50-100 ha in size and of the order of 5-15 m in depth. The application technique has generally

FIGURE 2.2.1:   Typical Design for Alum Addition (after Kennedy and Cooke, 1982)
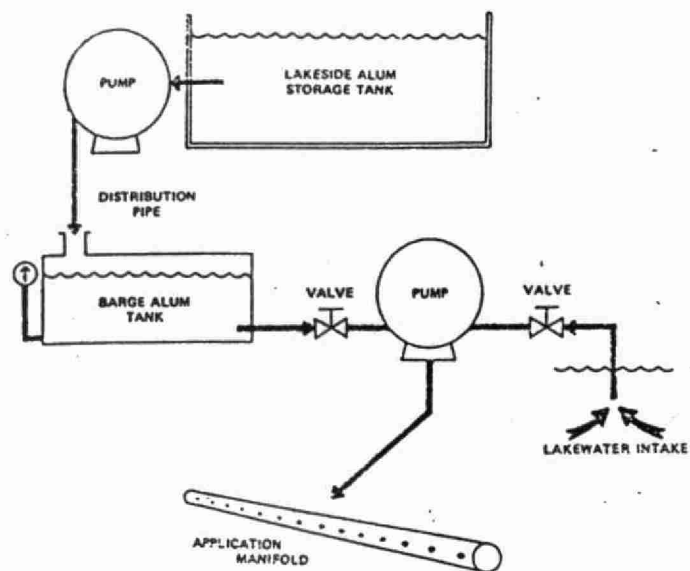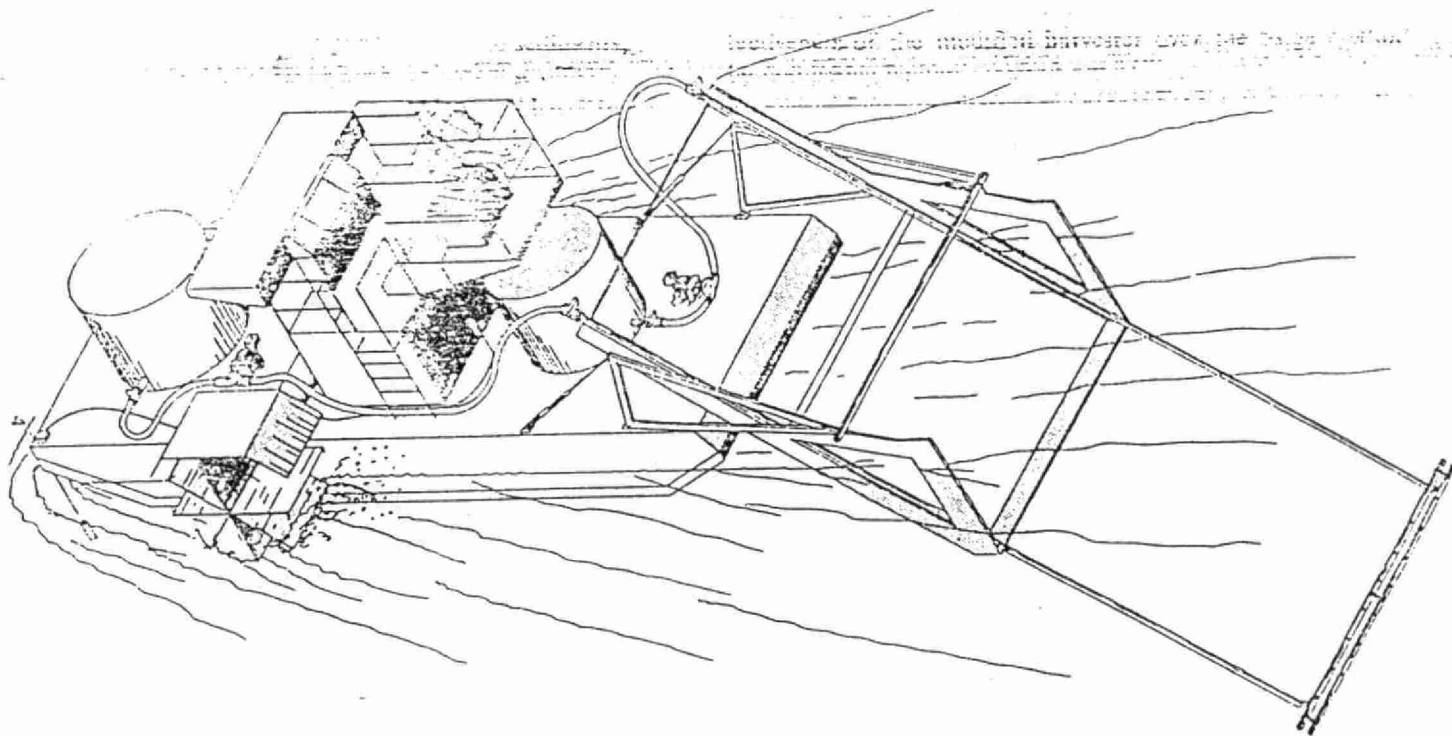
FIGURE 2.2.2:   Sketch of Alum/Sodium Aluminate Treatment Apparatus

consisted of either surface application, metalimnetic application, or hypolimnetic application. In all cases, the application has been made from the application boom lowered to the depth of application. This is, hence, a plausible technique for application to lakes with shallow depths. However, scale-up of this technique to much deeper lakes, (for example 20-25 m) is difficult and would require more design work for evaluation of its application. In the case of the Bay of Quinte which has a depth of the order of 5 m, existing application techniques should be sufficient to allow one to design an adequate system for application to the Bay of Quinte. However, the major differences would be that most of these lakes are of the order of 50-100 ha which is approximately 5-10% of the surface area of the Upper Bay of Quinte which would need to be treated.

2.2b   Iron Salts

Iron salts have been applied much less frequently to lakes for purposes of phosphorus inactivation than aluminum salts.

Two approaches have been used. One involved injection of ferric salts to the epilimnion, the metalimnion, or the hypolimnion. Techniques used are similar to those for alum addition. Literature to date indicate the addition of ferrous ion, in conjunction with aluminum into the metalimnion (as ferrous aluminum sulphate) (Roy, 1985) and of ferric alum blocks (May and Baker, 1978) into ponds have been successful in retarding internal P loading. The addition of ferrous iron in the form of pickling liquor has been proposed for Wood Lake in the Okanagan Valley, but not implemented pending evaluation of costs and approval by appropriate government authorities (Murphy, personal communication).

The second method involves injection of ferrous sulphate to the river water flowing into the water body (Hockstra and Malwald, 1985). The technique was applied to a Dutch lake (Andelsee Mass) which is used as a water supply source. Algal blooms were controlled to the range of 25 to 50 ug/L of chlorophyll-$\underline{a}$ for purposes of water tratment with addition of 4 tonne/day (iron-II-sulphate) to the influent stream whose flow rate was approximately 3 cms). The Andelsee Mass basin (a shallow basin, volume = $9 \times 10^6 m^3$) is a reasonable size for evaluating the requirements of the Upper Bay of Quinte (surface area = 136 km^2, volume = $476 \times 10^6 m^3$), being approximately one-tenth the size of the upper Bay of Quinte. The dosage requirements are complicated by hydrodynamics which are subjected to dispersion due to tidal influences.

2.2c   Calcium Salts

Calcium salts have been considered for application less frequently than alum or iron salts.   Calcium salts have been considered more actively for neutralization of acid precipitation and its effects upon lakes of low alkalinity (e.g. Sverdrup, 1984).   A listing of a few studies which have applied lime for control of algal composition and internal loading is given in Table 2.2.1.

The idea for lime addition for control of algae and internal loading has been advanced by workers such as Birdsey et al. (1984) and Murphy et al. (1983) who have empirically noted the disappearance of phosphorus from the water column in association with natural processes (elevated pH during algal photosynthesis) which induced precipitation of calcium carbonate.   The fundamentals of these processes have generally not been evaluated in lake rehabilitation studies oriented towards internal loading control to the same extent as other research studies (e.g. Stabel, 1985), which have examined naturally induced precipitation processes.

The method of treatment is to add lime (calcium carbonate, or slaked lime, calcium hydroxide) to the water from a boat or barge; the device used by Prepas and Murphy (1987) is shown in Figure 2.2.3.   Alternative methods include Swedish technology for lime addition to small acid lakes; these techniques are generally quite advanced but are similar in principal to those of Figure 2.2.1.   The main objective of Prepas and Murphy was to add lime to the water rather than to the sediments. This has direct effects on the algal community.   The main effect in many studies including that of Prepas and Murphy, however, appears to be to precipitate, carbonate which settles to the sediments. Lime addition to sediments is effective for a period of time, but it becomes inactive with time.

2.3   Description of Biochemical Oxidation

Biochemical oxidation of reduced lake sediments in situ with nitrate was proposed by Ripl (1978).   The method is based on microbial denitrification in which nitrate is utilized as a terminal electron acceptor, rather than oxygen, by certain bacterial taxa at low oxygen concentrations.   Nitrogen is released to the atmosphere as N_2. The advantages of biochemical oxidation are: 1) higher redox values in surficial sediments which retards P

TABLE 2.2.1:   TREATMENTS WITH LIME

Researchers Lakes	Form	Dosage	Purpose
Johnson and Hasler 1954 Elser et al. 1986 Peter and Paul Lakes	$Ca(OH)_2$	0.1 to 24	Improve water clarity
Murphy, pers. comm Hamilton Harbour	$Ca(OH)_2$	(560 g/m^2) 28	Remove indutrial contaminants
Murphy et al. 1985 Frisken Lake	$Ca(OH)_2$	max. 8.6	Reduce phosphorus and chlorophyll levels
Bengtsson et al. 1980	$CaCO_3$	10 to 20	Increase pH
Prepas and Murphy, 1986	$CaCO_3$	(43 g/m^2) 14 mg/L	Decrease algal biomass
	$Ca(OH)_2$	(41 g/m^2) 13.2 mg/L	Control winter kill

FIGURE 2.2.3:   CaCO$_3$ Applicatin Craft
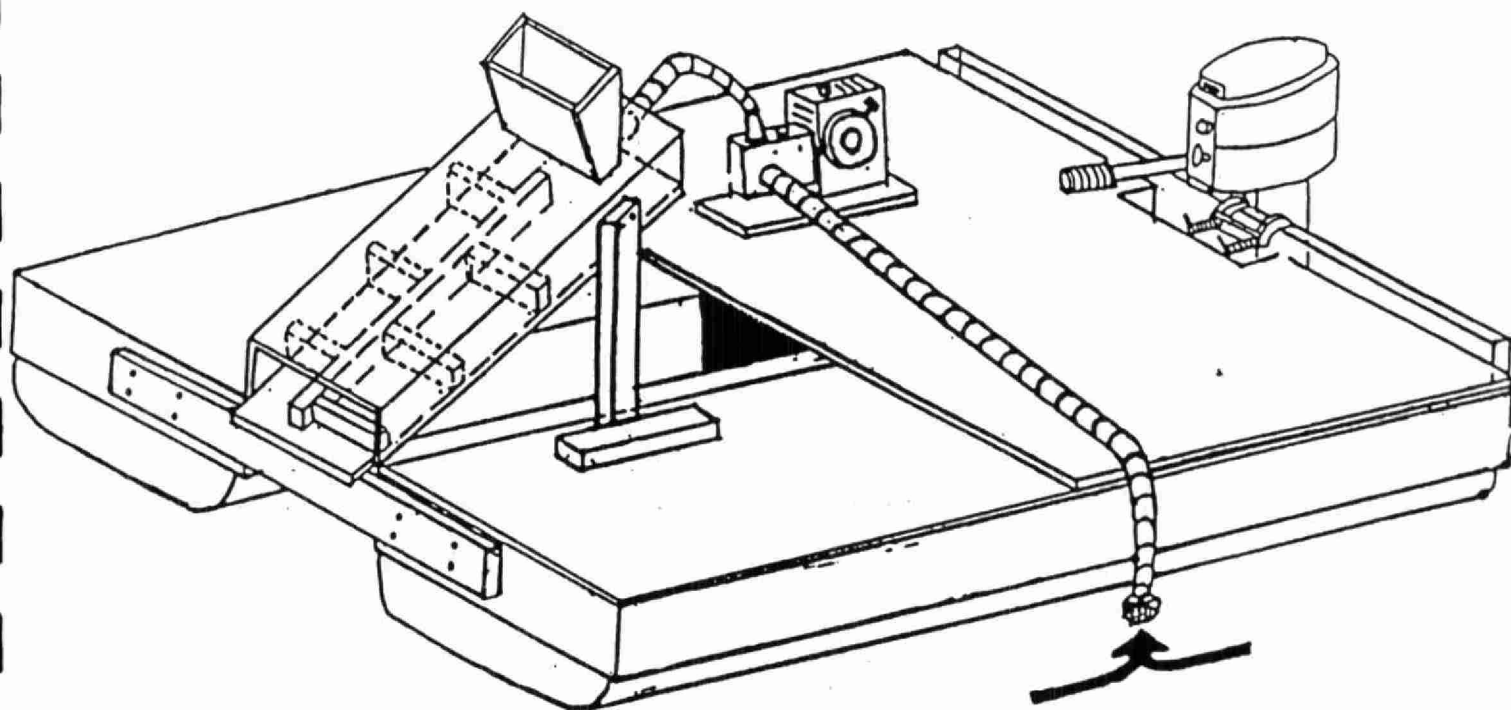
release (Ripl and Lindmark, 1978; Willenbring et al., 1984; Foy, 1986), 2) accelerated decomposition of organic matter which reduces dredging requirements (Ripl, 1985; Verner, 1985); 3) decreased chemical oxygen demand (Ripl, 1978; Ripl and Lindmark, 1978); and 4) decreased blue-green algal dominance (Ripl, 1985; McQueen and Lean, 1987), possibly due to increased CO_2 production accompanying accelerated decomposition (Molot, 1988).

Two nitrate application methods are commercially available. The RIPLOX method involves application of commercially available liquid calcium nitrate via a modified agricultural harrow directly into lake sediments (Ripl 1978; Willenbring et al., 1984; Verner, 1985). Addition of $FeCl_3$ may be necessary to enhance immobilization of sediment P (Willenbring et al., 1984). Treatment devices can be obtained from Atlas Copco Aquatec (70 Demarest Drive, Wayne, New Jersey 07470, U.S.A. (201) 696-0554, attn: Richard S. Geney). A modification of this technique, in which solid calcium nitrate is used instead of liquid, may be desirable because of rapid flushing in the Bay of Quinte (Minns et al., 1986). It is likely that annual applications will be necessary, probably in early spring at the beginning of the growing season. Application should be unnecessary during the winter when sediments are aerobic.

Nitrification steps have also been proposed for sewage treatment plants (Ripl, 1985; Ripl, 1986). This method, in which NH_3 is converted to nitrate prior to discharge of treated sewage into receiving waters, has the advantage of further reducing oxygen demand because of the reduced NH_3 load. Hence, the required nitrate dosage would be less if the STP nitrification step was used rather than the RIPLOX method.

The choice of technique will depend upon the required dosage, the quantity of point-source NH_3 available for conversion to nitrate, cost of implementation and annual operation, environmental impact, and finally, the effects of rapid flushing in the Bay of Quinte on discrete dosing (RIPLOX) and continuous dosing (STP nitrification).

Dosage will depend upon the increase in E_h necessary to retard P release and can be estimated by converting sediment oxygen demand (SOD) to a nitrate demand, after correction for reduced NH_3 loading if STP nitrification is implemented. Alternatively, dosage can be calculated if the labile organic matter and NH_3 loads from in situ, point and non-point sources are known. In either case, molar conversion ratios of 2:1 for O_2:N and 1:1 for C:N can be used.

2.4 Description of Fly Ash Application

This method involves application of power plant fly ash to eutrophic lake sediments (Theis and McCabe, 1978; Theis et al., 1979). Fly ash is a fine grained, largely inorganic residue remaining after combustion of coal and is largely a waste product which varies in composition. The physical and chemical properties of fly ash depend on the composition of coal burned, type of combustion and ash collection equipment employed, and method of storage of disposal.

The application of fly ash and lime to a depth of 4-5 cm in parts of Lake Charles East effectively sealed the sediments and reduced (but did not eliminate) sediment P release from treated sediments during anoxic periods (Theis and McCabe, 1978; Theis et al., 1979). The effective mechanisms are postulated to be an increase in the diffusional path length and an unknown chemical reaction (precipitation or adsorption). Furthermore, SOD probably decreased, as evidenced by higher nitrate levels in the water column after treatment. Interpretation of nitrate data is complicated by a 90% reduction in P loading to the lake the year prior to fly ash application.

Theis and McCabe (1978) recommended that fly ash with relatively low amounts of nitrate-dithionite-bicarbonate extractable Fe, and relatively large amounts of lime be used. The fly ash should not sink below the sediment/water interface. Continual deposition of organic matter above a fly ash layer necessitates fly ash application at regular intervals, possibly as frequently as annually. Detailed models are required to estimate frequency of application.

2.5 Description of Barriers

A   Curtains and Booms

A barrier such as curtain could be placed across the Upper Bay of Quinte to decrease internal loading induced by currents. These loadings would be those associated with erosion of particulate material which subsequently releases phosphorus in the water column and velocity gradient effects upon diffusional transport. The effectiveness of the barriers upon internal loading can be only be approximated at the moment because the degree for which internal loading is controlled by currents is unclear. Certain recent

research on Dutch Lakes (e.g., Aalderink et al., 1985) allow some insights into the phenomonen.

Barriers may consist of curtain placed throughout the water column and across the Bay to minimize current velocities. Several barriers would be required to adequately minimize currents in the whole bay. Alternatively, a limited number of curtains could be placed to minimize internal loading in particular portions of the Bay. However, no literature was found which used barriers for purposes of minimizing internal loading. Rather, such barriers have been used for protecting swimming areas from certain biota (e.g., Valens Reservoir, near Hamilton, Ontario) and for limnocorrals which are enclosures containing lake water and which are experimentally manipulated.

To determine whether only local areas or whether the whole lake should be included in a barrier scheme, phosphorus data for the sediments and phosphorus release estimates being measured by BEAK, were evaluated. The data of Johnson (Table 1.5.1) do not show a substantial enrichment of phosphorus in any particular area. The phosphorus flux data are not adequate to permit us to determine whether curent-induced release, decomposition-induced release and diffusion-control are the main factors in internal loading. Accordingly, it is not possible to establish with adssuredness, which areas should be considered for placing curtains. In this study, assumptions are made with respect to placement of curtains across the Bay. Evaluation of their potential effectiveness and of whether this spacing is optimal, awaits future evaluation.

B   Macrophyte Beds

Traditionally, macrophytes have been evaluated in the lake restoration literature (e.g., Cooke et al., 1986) in terms of plant harvesting. The broadly interpreted objective is to remove unwanted plant or plant parts to permit the desired use of the water of the littoral zone. In wastewater treatment (e.g., lagoons), harvesting of such underwater plants as Hyacinth and Duckweed have been implemented as methods for phosphorus removal (e.g., Middlebrooks et al., 1982) and/or for providing a food source for chickens in developing countries. Harvesting is inappropriate in the Bay of Quinte because, in the currently stressed ecosystem, existing beds are dying, resulting in a loss of fisheries habitat - the loss of the macrophyte allows sediment erosion by currents and disturbance of stone beds where eggs are laid, and possibly increases the internal loading.

In this report, the possibility for increasing the area of macrophyte beds are mentioned as one possible method for controlling internal loading. However, the logistics and costs of implementing such controls are not evaluated because it is not clear why areas previously covered by macrophytes have lost their plants. Ideas hypothesized by others include excessive phosphorus levels, competition for nutrients with floating plants, man-caused physical disturbances, and changes in predators. Because the success of establishing beds and the frequency of replacing them cannot be realistically evaluated (i.e., once every two years, once every decade), it is difficult to develop reasonable cost estimates. Furthermore, macrophyte die-off may contribute to internal loading.

2.6   Description of Dredging

There are two main steps involved in dredging for control of internal loading. They are the dredging activity itself in which sediments are removed from the bottom, and secondly, the disposal of the dredge spoil. Disposal areas include open water, a confined area in which the disposal area is isolated from the rest of the water body by a dyke, and thirdly, disposal on land.

Dredging of isolated areas of the Upper Bay of Quinte sediments may be a cost-effective option for control of internal loading. However, dredging of the whole Upper Bay of Quinte is also critiqued. For disposal, the impact of disposal to an underwater environment or to a land facility are critiqued.

The two main dredging techniques are: suction, and dredging with a bucket. In the case of suction, a barge is anchored near the site to be dredged and a pipe with suitable head inserted into the sediments. Water is pumped under pressure from the sediments and then either piped to a disposal area with suitable piping arrangements or loaded onto a second barge which can store the material and transport it for disposal. The pumping action sucks sediments from the bottom at the same time as water thus creating a slurry of sediments and water for disposal. If the sediments are sufficiently cohesive, a cutterhead which erodes the sediments may be employed in conjunction with suction.

In the bucket approach, a boom and a dredge bucket are mounted on a barge and anchored in the area that sediment is to be dredged. Analogous to the way in which back-hoes are used on land to dig holes or trenches, the dredge with the bucket assembly

is used to dredge sediments. Generally, the bucket is lowered to the sediments through the water column, dragged to fill the bucket with sediment, brought back up through the water surface, and then the dredged sediments are deposited onto a second barge. This motion is repeated until the barge receiving the dredge spoil is full, at which time it is moved to the disposal site. There are different types of buckets available for use, including a conventional open clam shell bucket, a water-tight clam-shell dredge, and a dust pan style head.

In disposing of the dredge spoil, the method of choice has often been disposal under water due to the logistics of disposal. In the case of underwater disposal, the barge containing the dredge spoil is transported to a confined spoil disposal area through a gap in the barrier containing the enclosed water or to deeper water, it is opened up, and then the dredge spoil allowed to settle out of its middle. In the case of land disposal, the dredge spoil must be pumped from the barge onto the land facility. This necessitates a somewhat more involved set of equipment for disposal.

2.7   Description of Sediment Burial

Sediment burial involves application of sediments with clean fill, sand, clay or similar type of material to retard phosphorus migration from the sediments. Clean fill provides a cap over the sediments to retard internal loading. These materials may be a thick barrier to retard migration by increasing the diffusion length from the contaminated sediments (such as putting several centimeters to meters of clean fill over top of the sediments) or simply may be a polyethylene type of membrane over the sediments to retard migration. In addition, the barrier may use geological materials which also absorb phosphorus to provide an additional mechanism for retarding migration in addition to increasing the diffusional length.

There have been no articles found in the literature in which sediment burial with clean fill was used as a technique for controlling internal loading. However, application of fly ash (Section 2.4) is in essence a sediment burial technique which uses a particular waste product as material to bury the sediments. Capping of contaminated dredge material with clean fill at an underwater site is similar to sediment burial. Placement of membranes composed of polyethylene, hypalon, polyvinyl (PVC), polypropylene, fiberglass screens (aqua screen coated with polyvinyl chloride), Dart Tech (black pigmented nylon

material composed of 2 m thick sheets), and burlap which have been applied to small areas in lakes to control macrophyte growth (Cooke et al., 1986) are also similar to the technique required for controlling internal loading. These techniques provide background for evaluating the possibilities for controlling internal loading in the Upper Bay of Quinte using sediment burial control.

Particular attributes for a membrane seen as being ideal as sediment cover for controlling macrophytes include the following (Perkins, 1984):

1. Sufficiently opaque to block photosynthetically active radiation;
2. Durable during and subsequent to application;
3. Negatively buoyant to aid in installation and to impede lifting or "ballooning";
4. Vents to allow escape of gas;
5. Smooth on upper surface, to inhibit fragment rooting; and
6. Competitively priced.

As Cooke et al., (1986) note, there is no material on the market that has all such attributes. In addition, for the purposes of controlling internal loading, all of these particular attributes would not necessary be most appropriate. For example, vented to allow the escale of gases would also allow diffusion of phosphorus. Accordingly, in this work, it is assumed that the sediment burial technique would be similar to the application of fly ash that an inert material such as clean fill would be used to cover the sediments. Further, it was decided that the techniques used for placement of dredged material and capping of dredged spoil as practised in various marine dredging applciations would be an appropriate scheme for evaluating the possibilities of burying sediments. Accordingly, these assumptions are used in developing and describing the logistics and costs of sediment burial.

2.8   Description of Artificial Circulation

2.8.1   Enhanced Circulation

This technique involves the use of mechanial energy to enhance circulation in a water body, thereby increasing both the rate of aeration at the water/atmosphere interface and the rate of oxygen supply to anoxic bottom waters and sediments. Enhanced circulation

differs from hypolimnetic aeration in that the latter relies on aeration of bottom waters without destratification. The primary function of enhanced circulation in the well mixed Upper Bay would be to maintain an oxidized microzone in surficial sediments.

Circulation may be enhanced by air lift pumps in which compressed air is injected into deep waters at a rate sufficient to cause turbulent mixing of lake waters with aeration taking place at the lake surface. The amount of water flow enduced by a rising bubble plume is primarily a function of air release depth and air flow rate (compressor size) (Lorenzen and Fast, 1977). Hence, this technique would be of limited use in the Upper Bay because of its shallow depth of water (Johnson, 1984). Operation would be continuous from April - November.

Enhanced circulation successfully reduced internal P loading in several studies - Heart Lake (Nalewajko et al., 1981; Nicholls et al., 1980; Molot, 1988) and Kettle Lake in Ontario (Ellis and Tait, 1982), and Lake Cachuma in California (Boehmke, 1984). Sediment P release may still occur under oxic conditions due to macrophyte die-off, turbulence-mediated resuspension, enhanced macroinvertebate activity and insufficient Fe for inhibition of diffusion (Pastorok et al., 1982).

2.8.2   Aeration

Aeration attempts to oxygenate the hypolimnion without destratifying a water body. The primary function is to keep the water adjacent to the sediments well oxygenated such that a well-oxidized surficial sediment zone is maintained.

Aeration, if carried out by air addition requires compressors which compress the air, a delivery line and a set of diffusers. If carried out by pure oxygen addition, a generation system on site or a transport system from an available source would be required, together with a series of storage tanks located on shore around the pheriphery of the Bay; each storage tank would supply a diffuser or a cluster of diffusers out in the bay. An alternative design would involve having a generation plant on shore, transfer of liquid oxygen to a supply barge, a series of storage tanks on barges anchored on the Bay from which oxygen is injected into the water, and the supply barge moving from storage tank to storage tank replenishing it.

Similar to enhanced circulation using air lift pumps, it is doubtful that aeration would be very effective due to the shallow depth and the generally well oxygenated conditions observed throughout the Upper Bay of Quinte. In order to ensure adequate oxygenation of the water body, several parameters must be estimated including the velocity of the water and the rate of respiration in the water. This data is necessary in order to size the appropriate aeration equipment. However, this data is not available in the literature and many assumptions must be made (Section 3.8.2).

2.9   Lake Drawdown

Lake drawdown is a technique suggested for application to various lakes and reservoirs to prevent or minimize hypolimnetic oxygen depletion and to induce the formation of aerobic sediments where the sediments are exposed to the atmosphere. The maintenance of oxygenated conditions prevents the anoxic release of phosphorus - a rate of internal loading which is often ten times or higher the rate of internal loading during aerobic (oxic) conditions.

Drawdown for the Bay of Quinte is infeasible. The inter-relationships of the water levels of the Bay of Quinte and downstream water bodies prevent much alteration to the water levels of the Bay of Quinte. The water levels of the Upper Bay, the Middle Bay, and the Lower Bay are controlled by water levels of Lake Ontario. During spring periods of high flow rates from the Napanee River, the Moira River, and the Trent River, the water levels of the Upper Bay are the same as the lake (Freeman and Prinsenberg, 1986). The largest cause of bay levels being different from lake levels would be wind driven currents. Level variations should be only a fraction of a millimeter or centimeter depending on the strength of the wind; these differences would decrease to nil upon relaxation of the wind.

It is plausible that an aeration dam could be placed across the Bay between the Upper and Middle Bay (see Figure 1.5.1) to separate the two water bodies and that the Upper Bay could be pumped down to lower its water level. This option would have an excessive social impact on communities such as Trenton, Belleville, Deseronto and Napanee, in addition to a substantive environmental impact upon fisheries, wildlife, etc.

Accordingly, drawdown is only mentioned in this report, but not considered further as a control option.

3.0   LOGISTICS OF IMPLEMENTATION

A detailed analysis of the logistics of implementation of each control option are given below.   Available literature has been used as a case history basis for documenting the logistics of implementation of each option. The analysis includes the following:

- o   materials procurement,
- o   method of implementation,
- o   time required (once, or several times over several years), and
- o   manpower requirements, operation and maintenance requirements.

The following headings are used for describing logistics of implementation.

3.1	Null Option - Natural Clean-up		
3.2	Chemical	3.2A	Alum Salts
		3.2B	Ferric Salts
		3.2C	Calcium
3.3	Biochemical Control		
3.4	Fly Ash		
3.5	Barriers		
3.6	Dredging		
3.7	Sediment Burial		
3.8	Aeration		
3.9	Lake Drawdown		

3.1   Null Option - Natural Clean-up

There are no direct questions concerning feasibility of the null control option to be addressed because it will occur naturally.

Because reduction of internal loading should result from control of point/diffuse sources, their feasibility will influence natural clean-up. Evaluation of their feasibility is given in an allied report (Canviro, 1987; Ecologistics, 1987).

3.2 Logistics of Application of Chemical Control Options

3.2a   Logistics of Application of Aluminum Salts

The application of alum to the Bay of Quinte requires first an estimate of the dosing requirements and secondly, an assessment of the method of implementation, of time required and manpower requirements.

For dosing requirements, it is dependent upon whether the objective is to strip phosphorus from the water column or whether the objective is to provide sufficient alum at the sediment-water interface to retard and minimize internal return of phosphorus over the long term.   In the case of Bull Head Lake, application of alum to the metalimnion showed that flocs were rapidly formed, that they settled to the sediments rapidly and that they were incorporated into the top 5 cm within a month, being concentrated at the 2 to 5 cm interval within the sediments. At the same time, there was minimal change in the total phosphorus concentration in the water column at the time of addition. This lack of removal of phosphorus from the water column must be attributed to the rapid kinetics of aluminum conversion into aluminum hydroxide which did not emmesh or adsorb very much phosphorus contained in the water column. Operating the aluminum addition at a different concentration point, as has been done successfully in wastewater treatment, would also strip phosphorus from the water column, but the necessity for this must be established in any particular application.

The main effect of the addition of the alum in the Bull Head Lake case was to begin reduction of internal loading of phosphorus within a two month period. Over the next two or three years, the internal loading was substantively reduced.   Control of phosphorus in the water column could be achieved by additional amounts of aluminum being added to the water column; however, the necessity for this in the Bay of Quinte case would need to be established because of the continuing loading of phosphorus to the lake from land based sources. It would seem, based on the direction of research and applications from the literature, that the main objective of alum addition should be directed towards controling internal loading as opposed to also removing phosphorus from the water column at time of application. This is particularly an appropriate objective for alum application to the Upper Bay of Quinte because it has a short residence time; this means that external loadings rapidly replenish the observed bay concentrations.

Kennedy and Cooke (1982) outlined a procedure for estimating dosing requirements. Briefly, it is as follows:

1. Obtain water samples and determine the alkalinity and pH of each sample.

2. From these values obtain initial estimates of the optimum dose for alum addition from a chart that they give (Figure 3.2.1).

3. Conduct a set of jar tests to determine that the dosages are generally within the optimal range for purposes of the formation of an aluminum hydroxide floc, its precipitation and settling to the sediments.

With respect to materials procurement, the four main forms of aluminum which can be used are liquid alum, granular alum to be mixed up and dissolved, sodium aluminate, and aluminum hydroxide gels. In the case of the first three forms, aluminum is dissolved in an aliquot of lake water and applied, whereas the aluminum hydroxide gels are deposited upon the sediments directly. Few field evaluations of addition of aluminum hydroxide gels have been reported. Accordingly, their method of application is not considered.

Another form of aluminum which has not been explored in the lake rehabilitation literature is that of polyaluminum chloride. Polyaluminum chloride (PACl) is presently being developed as a substitute for alum in water treatment practice. For coagulation processes, it has the advantage of requiring a lower dose of aluminum contained in the PACl solution than alum to provide effective removal of suspended solids and other contaminants within the water treatment plant. It is, however, more expensive, and the aluminum within it is dominated by a polymeric form of aluminum which is stable for long periods of time. Such stability may preclude the formation of aluminum hydroxide floc depending upon its dosage and other factors. Also, the effectiveness of PACl for phosphorus control is not clear given its polymeric structure, and hence its potential application is not evaluated.

For purposes of this report, it is assumed that the addition of alum would be the preferred form for aluminum addition because of its readily availability from commercial suppliers who also supply alum for coagulation processes in water treatment, and for chemical precipitation in wastewater treatment. It is assumed that liquid alum would be

4111.1   3.3

FIGURE 3.2.1:   Optimum Dose for Alum Addition
(after Kennedy and Cooke, 1982)

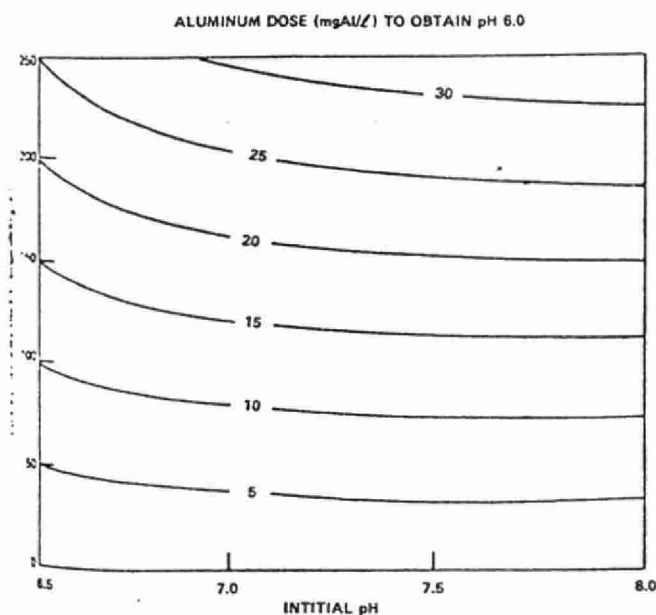ALUMINUM DOSE (mgAl/ℓ) TO OBTAIN pH 6.0

INTITIAL pH

Figure 5. Estimated Aluminum Sulfate Dose (mg Al/ℓ) Required to
Obtain pH 6 in Treated Water of Varying Initial Alkalinity and
pH. Based on equations in Ferguson and King (1977) and
assuming insignificant phosphorus concentrations.

delivered to a shore facility located near one of the sewage treatment plants in 6,000 gallon trucks, and that this material would be pumped from the truck onto a barge for application directly to the lake.

Of the methods described in the literature for application of alum to a lake, the method of Connor and Smith (1986) is taken in this work as being a reasonable approach and hence, used for purposes of describing the logistics of application and costing. Application consists of the following steps. The tanks on the barge are filled with alum and then the field crew makes trips of approximately 45 minutes with the harvester out into the lake to inject the alum before returning to shore to refill the chemical tanks. Three men working on board the harvester were required to apply and regulate the chemicals during the application. A third crew of 2 people was used to move buoys from location to location to ensure that all the specified areas were treated. Problems could occur with wind affecting the course of the harvester, but in the application involved, Connor and Smith stated that even occasional gusts of wind did not significantly inhibit the application, albeit they do note that the weather was fair with little wind in general. They required approximately 6 hours of manpower each day for two days to accomplish application to the 10 hectares. Scaling this number up then, for 13,600 hectares of the Bay of Quinte, approximatley 8100 crew-days would be required in order to apply the alum using its harvester. This is clearly unfeasible for application to the scale of the Bay of Quinte, and hence, more barges and possibly a larger design would be required to make the application effective. Assuming the efficiencies and time and other related matters, if 100 of the spreaders were used, then approximately 80 days would be required for application of the alum to the Bay of Quinte.

Hence, it is apparent that a larger device than that used by Connor and Smith and others (who have used a similar size of barge) would be required to make application of this technique feasible for the Bay of Quinte. In this work, such a device is not designed. But this size of device is retained for costing all chemical control options because costing information has been developed for these sizes of devices. This maintains consistency of costing.

With respect to the number of times that alum would need to be added to the lake sediments, the literature does not allow us to make definitive statements. The frequency is a function of such factors as detention time, sedimentation rate and rate of aging or

coating of the flocs. In the Bay of Quinte, the rapid flushing time is a complicating factor for assessing the number of applications required because the applications described in the literature are generally on small lakes with presumably relatively longer detention times (of the order of a year or longer). In particular, several descriptions of applications do not note such data as detention time so it makes it difficult to give precise estimates of the number of applications required.

In the case of application of alum and aluminum to Bull Head Lake, it was noted that phosphorus concentrations in the hypolimnion were reduced approximately 70% and that phosphorus concentrations in the hypolimnion were reduced approximately 97% in the year following application. For the next four years for which data are reported, the rate of phosphorus recycling continued to be low hence, indicating that control was effective for at least a period of approximately five years. In another case (Welch et al., 1986) in which alum was added to a shallow lake (2 m mean depth) internal loading was reduced for four years but in the fifth summer after treatment, the lake returned to its pre-treatment levels. In evaluating the application to Bull Head Lake, depth inspection in the sediments noted that the alum was located 2-4 cm deep in the sediments. Accordingly, this would act as a barrier for control; any additional recycling of phosphorus from the sediments are presumed to result from new inputs of phosphorus to the sediments which then recycle back from surficial sediments located at depths above the alum lens within the sediment structure.

Accordingly, for the Bay of Quinte, precise estimates of the frequency with which alum treatments are unclear. In this study, it is assumed that application every three years would be necessary to minimize the release of phosphorus in the sediments based on the apparent trends within Long Lake (Welch et al., 1986) and within Bull Head Lake (Nart, 1985).

3.2b   Logistics of Implementation of Iron Addition

The logistics of applying iron are similar to those of alum. It is assumed that the best method for application is direct injection into the water overlying the sediments, rather than into a riverine discharge (eg., similar to Hoekstra and Maiwald, 1985), although this has not been directly established.

Accordingly, methods of application of iron would be similar to those of alum (e.g., similar widths of application, concentrations of stock solution, similar equipment). Somewhat different dosages may be appropriate; however, further experimental analysis in the laboratory and field is required to establish precise differences. This is beyond the scope of this project.

3.2c   Logistics of Implementation of Calcium Salts Addition

There are two sets of data which can be used to evaluate alternatives for application: application of calcium salts to lakes to induce phosphorus precipitation and addition of calcium salts to acid lakes to neutralize the lake water and sediments. A synopsis of application methods is given in Figure 3.2.2. Truck transport may be feasible for the Bay of Quinte if the ice is thick enough. In this method, lime or limestone is distributed onto ice; upon ice melting, the calcium salts dissolve in the water. Only a portion dissolve in hard water, with the remainder settling to the sediment.

In the work of Prepas and Murphy (1987), a device similar to a weed harvestor containing an apparatus for slurrying lime (Figure 2.2.3) was used on a 32 ha lake. "Two types" of lime were used - calcium carbonate and calcium hydroxide. The lime (approximately 15 tonnes) was trucked in bags from Edmonton. The lime was unloaded from the trucks by fork lift and carried to floating docks from which it was manually unloaded onto the barge into 1 tonne lots. The barge was a 4.8 by 3.3 m deck fixed onto two epifoam pontoons and powered by a 9.9 hp. outboard motor (Figure 2.2.3). The slurry maker was a 2.4 x 1.2 x 10.25 m plywood box which enclosed a series of baffles. The slurry assembly was placed at a 45° tilt with its open end projecting down over the bow of the barge. Bags of lime were cut open and poured into a slot located at the height of the slurry maker. Water was injected to one end of the slurry maker with a 7.62 cm outlet water pump, and left through the other end of the slurry maker which was left open. Water from the 7.62 cm outlet pump and lime were thoroughly mixed as both substances cascaded over the series of baffles. This type of slurrying device is unlikely to efficiently separate particles of lime which adhere to each other in a dry state because of electrostatic forces. For acid lakes, separation of fine particles is a prime objective. But, for control of internal loading, getting the particles to the lake sediment with minimal dissolution may be a desired objective.

FIGURE 3.2.2:   Synopsis of Calcium Salt Application
                  (after Cooke et. al., 1986)
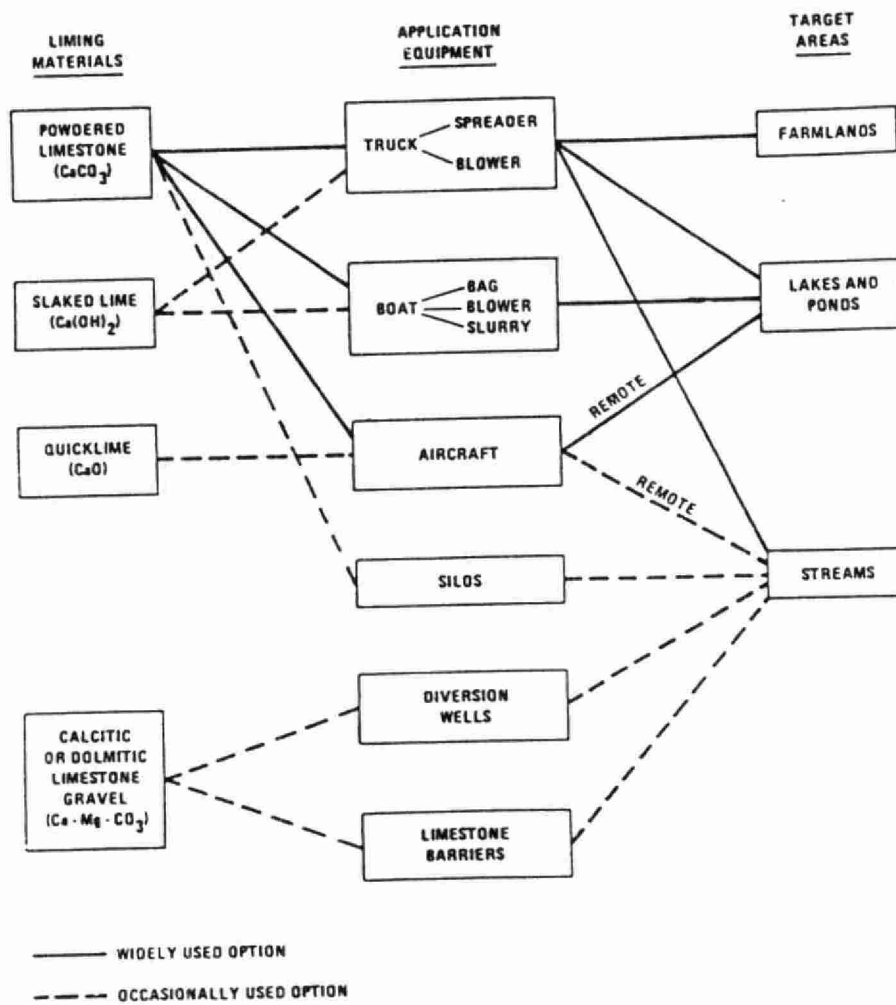
A four person crew was used to apply the lime to the lake. Thirty minutes were required per trip. Personnel on board the barge wore arm length rubber gloves, rain suits, rubber boots, and dust masks. Lime handlers on shore usually did not wear full protective equipment.

For purposes of this work, it is assumed that an apparatus similar to the weed harvester for alum would be used for application of lime to be discharged close to the sediment-water injection. A slaking system similar to that of Prepas and Murphy (1987) would be required. Injection near the sediment-water interface rather than to surface waters would be used to maximize potential for long-term impact upon internal loading.

3.3 Logistics of Implementation of Biochemical Oxidation

There are two methods of application which influence logistical considerations. The first is the RIPLOX method which involves application of commercially available liquid calcium nitrate directly into sediments via a modified agricultural harrow. The equipment is described in detail by Kerner (1985). A variation of this method involves application of calcium nitrate pellets. This method of application is sketched in Figure 3.3.1. Detailed information regarding the RIPLOX method can be obtained from Atlas Copco Aquatec (70 Demarest Drive, Wayne, NJ, 07470, U.S.A., (201) 696-0554, attn: Richard S. Geney). Atlas Copco offers its services as a contractor, rather than as a distributor of equipment. Finally, nitrate can be added directly to STP effluent.

The nitrate load should equal the SOD to effectively reduce internal phosphorus loading. The estimated SOD for the Upper Bay of Quinte is based on several assumptions. First, the sediment oxygen demand rate (SOD) has been estimated to be 0.22 g O_2/m^2/day and the areal hypolimnetic oxygen consumption rate (AHOCR 1.37 for the Lower Bay of Quinte at 10.8°C (Minns and Johnson, 1986). The ratio of SOD/AHOCR is 0.16, indicating that a large part of the decomposition of settling organic matter takes place in the water column above the sediments. SOD will be significantly higher in the Upper Bay during the summer because it is shallower, warmer and has a higher organic carbon loading. Therefore, it is assumed that the SOD is 1.0 g O_2/m^2/day in the Upper Bay (31 m moles/m^2/day). Hence, the required total nitrate load for the period May - October in the Upper Bay is 7800 tonnes NO_3-N or 57,000 tonnes of $Ca(NO_3)_2$.

FIGURE 3.3.1:   Conceptual Diagram of Equipment Designed to Distribute Calcium
Nitrate Pellets Over Lake Surface Waters.

The pellets are stored in a barge-mounted hopper and distributed by
means of a rotating auger while the barge moves over a targeted
area.
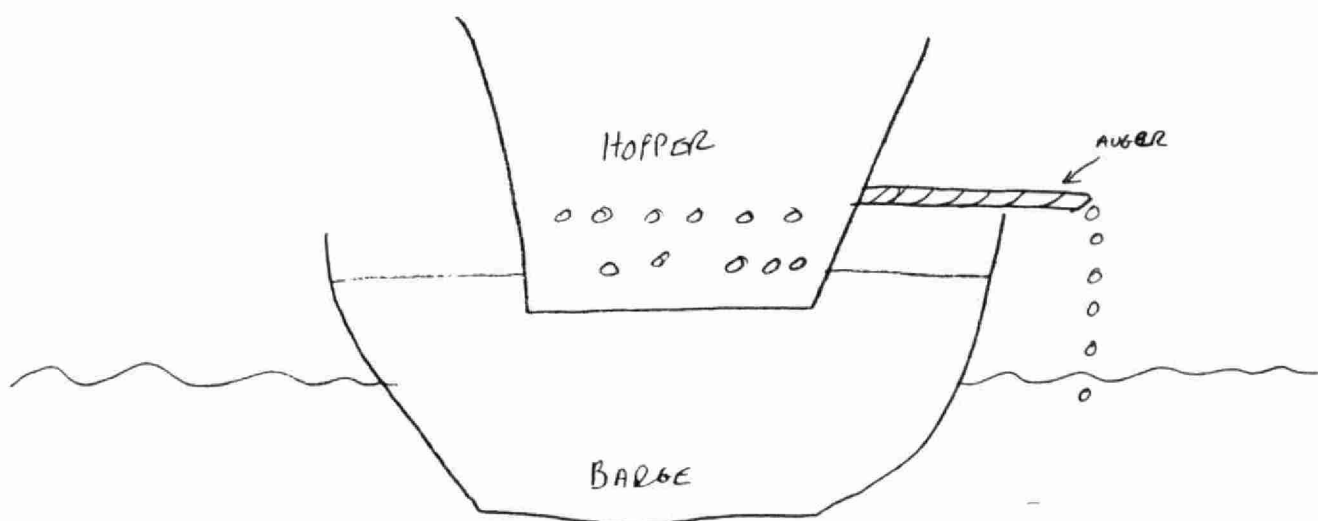
A correction factor for assimilation by algae and transport downstream should be adopted. For purposes of this report, the corrected dosage is assumed to be 100,000 tonnes of $Ca(NO_3)_2$ with the daily dosage being 560 tonnes of $Ca(NO_3)_2$ or 76 tonnes of NO_3-N.

Upstream addition of nitrate may be desirable. For example, addition of nitrate to Peterborough STP effluent or to Rice Lake directly may increase P retention upstream, thereby improving water quality upstream and reducing P export to the Bay of Quinte. The disadvantage of upstream nitrification is that failure to dose after many years of successful retention may result in a large release of sediment P. Furthermore, it must be demonstrated that redox mediated processes are primarily responsible for P release rather than annual dieback of the macrophyte Potamogeton crispus in Rice Lake.

The potential NO_3 load for the Upper Bay via STP nitrification is 432 Kg NO_3-N/day. This assumes that the average daily N load from point sources is 900 kg/day (Minns et al., 1987), that the average effluent compositions from an STP is 60% NH_3 and that the conversion efficiency for the nitrification process is 80%. Assuming a surface area of 136 km^2, the daily areal load will be 0.23 mmoles NO_3/m^2/day which is less than 1% of the estimated SOD.

The effectiveness of STP nitrification may be increased somewhat by including nitrification steps at upstream point sources such as the Peterborough STP. However, less than 10% of the potential SOD will be met. Hence, enrichment of STP effluent with $Ca(NO_3)_2$ is seen as the only viable means of achieving adequate biochemical oxidation.

3.4   Logistics of Implementation of Fly-Ash Application

Approximately 460 tonnes/ha is recommended for effective treatment (Theis et al., 1979). Hence, 6 million tonnes of fly ash would be needed to seal the Upper Bay of Quinte. Fly ash is readily available in southern Ontario from Ontario Hydro although it is not known if the composition is suitable or the amounts adequate.

This operation will require a large number of trucks, several storage sites along the shore of the Bay, heavy equipment for mixing the fly ash, and a method of application. The application method must be able to apply large amounts of fly ash in a short period of

time to form an intact layer, and must be able to apply fly ash over a large area in the Upper Bay.

The application technique described in Section 3.2c, the barge in Figure 3.3.1 or the Swedish MOVAB was developed to treat small, acidified lakes with calcite slurry. The MOVAB method consists of a boat capable of storing 7 tonnes of material, preparing a slurry and pumping the slurry overboard. The application rate is approximately 7 tonnes/quarter hour. Hence, one boat is capable of applying a maximum of 200 tonnes in an 8 hour work day, although 175 tonnes is more realistic. Therefore, 34,000 boat-day or 1500 boat-months are required to complete the operation. Each boat requires 2 operators on board and 2 people to operate the heavy equipment on shore and load the boats. Typically, material is pumped on board by means of a fluidizing technique. A navigation technique similar to that described in Section 3.7 is required to ensure even distribution of fly ash.

It is obvious that application methods developed for treating relatively small water bodies are unsuitable for use in the Bay of Quinte. An alternative application method, such as that described in Section 3.7 for sediment burial, may be more appropriate. Solutions to this engineering problem are beyond the scope of this report. For purposes of this report, application of the MOVAB method are costed for fly ash application.

3.5   Logistics of Emplacing Barriers

The logistics of placing barriers (curtains) is abstracted from the technique used by the Hamilton Regional Conservation Authority for protecting swimming areas from parasites involved in swimmers itch. The barrier consists of a curtain floated with a float-collar assembly and weighted down by a chain and anchors. The chain is run along about 0.6 meters from the bottom of the curtain and ordinarily sits upon the sediments. The chain is attached to the nylon curtain at pockets sewn into the nylon. Anchors are placed at intervals to maintain the position of the curtain. The bottom 0.6 m of the curtain normally sits on the sediments; however, if the surface of the lake rises, the floats raise the entire curtain. But because lake levels generally do not rise more than 0.6 m over the summer time, the curtain is always in contact with the sediments.

Another design which could be used is that of limnocorrals. A similar floating arrangement is used to support the curtain, but the curtain is typically made of plastic and its bottom is anchored and is placed into the sediments to prevent any water from moving into limnocoral. Whether it is necessary to place the curtain into the sediments to minimize lake currents, is unclear. In this work it is assumed that the HRCA design is adequate.

The logistics of placement require a three-man crew for about one day to place 400 m of curtain in the spring, and about three days in the fall to retrieve the curtain, clean it, and store it. Based upon operating history, the life-span of the curtain is approximately ten years. The logistics of placement involve working from a boat as the curtain is placed in up to 8-9 feet of water. The curtain must be unrolled, chain, floats and anchors are attached as it is unrolled, and then dropped into place as more curtain is unrolled from the boat.

For application to the Bay of Quinte, it is assumed for a generic assessment that the most effective use of a barrier would be in the shallowest portion of the Bay, i.e., between Trenton and Belleville. The Upper Bay of Quinte is 35 km long overall and has a mean depth of 3.5 m, a maximum depth of 14.9 m and generally varies between 4 and 8 m in depth. The distance between Trenton and Belleville is approximately 18 km with a width varying from approximately 1-3 km. For assessment of costs (next section), it is assumed that a curtain is placed at 0.5 km intervals across the Bay with an average depth of 3.5 m and an average length of 2 km.

If booms were placed rather than curtains, it is assumed that they would be placed only across the mouth of certain channels or around marinas. Similar logistics as laid out above are involved, but the scale would be much reduced depending upon the application.

3.6   Logistics of Implementation of Dredging

3.6.1   Logistics of Dredging

Traditional dredging practice has evolved to give the greatest possible economics (e.g. maximize dredged volume), but only secondary considerations have been given to environmental effects and aesthetic impacts. Conventional dredges are not intended to

remove contaminated materials resting upon the lake bottom. However, recent research has demonstrated that the main effect of dredging is to cause suspended solids plumes in the vicinity of the dredged area. This has led to new designs for dredges which have focussed upon minimizing residual suspended solids in the water column. In addition, the dredged area may be isolated with a curtain to minimize transport of the suspended sediment.

The conventional dredge, called a bucket dredge (see Figure 3.6.1), consists of a bucket operated from a crane or derrick mounted on a barge or on land. It is used extensively for removing relatively small volumes of material around docks and piers. The cutterhead dredge (see Figure 3.6.2) uses hydraulic suction to remove sediment and a cutter to loosen consolidated sediments. The main effect of the cutterhead is to loosen sediments and create a near-bottom plume. The main effect of the bucket dredge is caused by the impact, penetration, and withdrawl of the bucket from the bottom sediments; elevated suspended solids levels are also found near the bottom. Secondary losses of suspended solids from the bucket dredge occur as the bucket is raised through the water and spillage from the top and through jaws as the bucket emerges from the water.

The greatest reduction in resuspension came with use of the "water-tight" clamshell bucket (Montgomery, 1984). Reductions were especially noted in the upper water column. However, a direct comparison showed that the "water-tight" bucket increased resuspension near the bottom, probably because of the impact of the shock wave of water (induced by the downward motion of the bucket) upon the sediments (Montgomery, 1984).

Other special purpose dredges have been developed to maintain a high solids content or to minimize resuspension. Examples are summarized in Table 3.6.1. As Montgomery (1984) notes, these are intended for specialty applications, (e.g., dredging contaminated sediments) rather than for routine maintenance operations.

In this study, it is assumed that the cutterhead suction device would be used. Due to the large surface area of the Upper Bay of Quinte, modifications as outlined by Cooke et al. (1986) may be in order. However, costing used in this study is based upon smaller scale dredging using the cutterhead device. The critical costing factors for sediment dredging are the surface area and depth of sediment. Sediment data for phosphorus are not

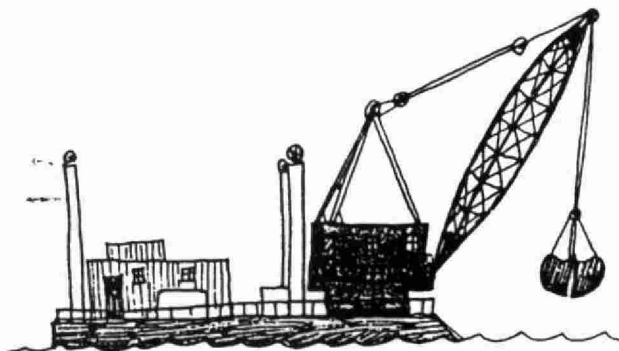FIGURE 3.6.1:   Sketch of Conventional Bucket Dredge
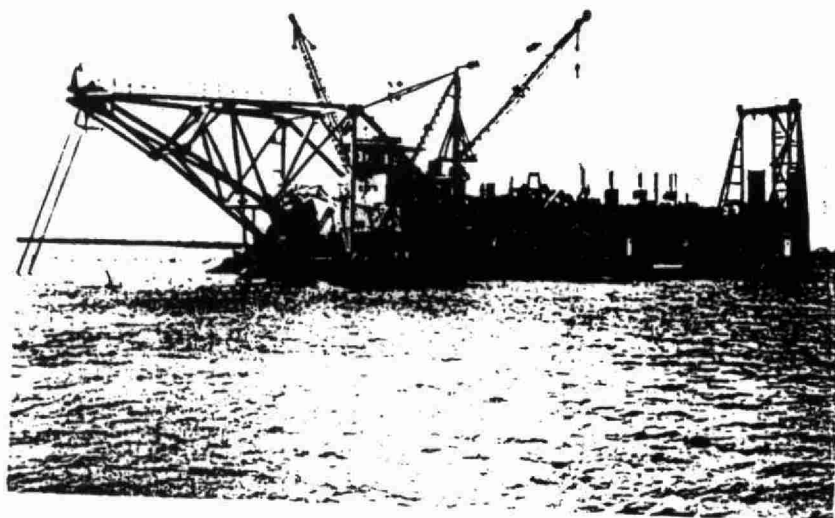

FIGURE 3.6.2:   Sketch of Hydraulic Suction Dredge

a.   Hydraulic cutterhead pipeline dredge (27 in.)

available for defining the depth of sediment required for dredging. Accordingly, it is assumed that 20 cm of sediment would be dredged.

3.6.2 Logistics of Disposal

Various designs for containment areas have been researched. They vary from having no cover to a clay type cap; having an underdrain system or no underdrain system, being overlain by bay water to being on land; having a treatment system or allowing leachate to migrate untreated. The required design depends upon location for disposal (on land or underwater) and type of contaminants in the sediments and their leachability.

The type of contaminants on Bay of Quinte sediments and their leachability has not been previously assessed. To be conservative, it is assumed that the following design is used:

o   land disposal, on the shore of the Bay of Quinte,

o   a sloping surface covered with soil to permit vegetation,

o   ditches around the disposal to collect external surface runoff,

o   a leachate collection system to collect surface runoff from the disposal area and infiltration.

The leachate will be routed through WWTP. Candidate sites for disposal are not evaluated in this generic assessment.

3.7   Logistics of Sediment Burial

Sketches of various aspects of disposal of dredge spoil and capping it underwater are shown in Figures 3.7.1, 3.7.2, 3.7.3 (Montgomery, 1986). In the initial dump (Figure 3.7.1), dredge spoil is barged to a particular area and dumped through a door in the floor of the barge to reach the bottom. This creates vortices and has various impacts on the water column. Next, during capping operations, a similar vehicle, but which is a barge under its own motor power and directed by surface buoys for location, is used to cap the sediments with clean fill. For dredge spoil, it is desirable to confine the dredge spoil disposal to as small an area as possible so as to minimize the aerial extent and amount of material (clean sand, etc.) required for capping purposes. An alternative placement technique is given in Figure 3.7.3 in which a submerged diffuser system is used to place
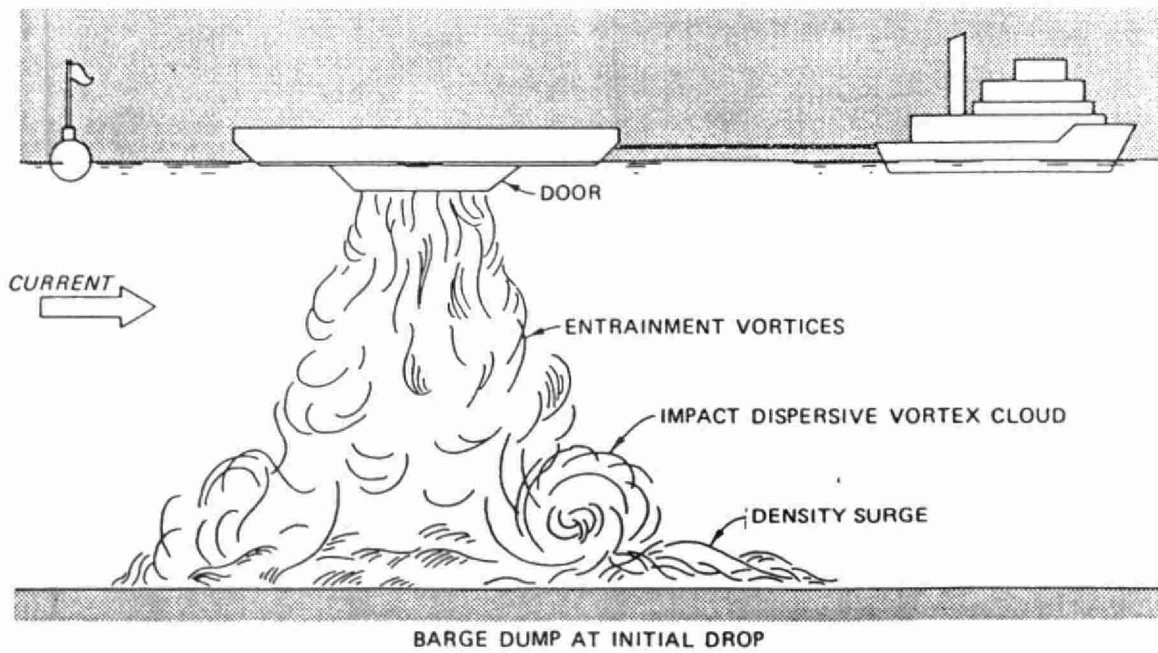
Barge Dump at Initial Drop

DOOR

CURRENT

ENTRAINMENT VORTICES

IMPACT DISPERSIVE VORTEX CLOUD

DENSITY SURGE

BARGE DUMP AT INITIAL DROP

FIGURE 3.7.1:   Dredged Material Discharged From a Barge/Scow

CAPPING SEDIMENTS
(CLEAN SAND, ETC.)

CONTAMINATED DREDGED MATERIAL

BOTTOM SEDIMENT

FIGURE 3.7.2:   Hopper Dredge Placing Cap Material

FIGURE 3.7.3:   Submerged Diffuser System
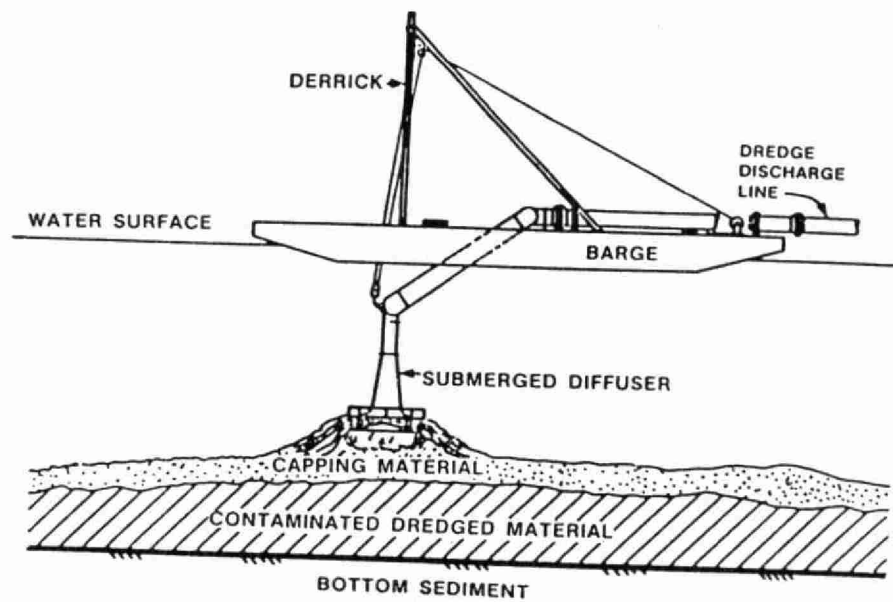
the capping material. In this case, the typical design uses a submerged diffuser which is able to place material to a fairly constant depth and with a fairly precise control on location. However, one drawback to this design is the supply pipeline required to the source of the material being disposed. This poses some logistical problems.

As Montgomery (1986) notes, this technique is particularly useful for minimizing the release of contaminants in sediments in the water column during placement; however, hydraulically placed sediments would spread over a much wider area than mechanically dredged and dumped sediments because of their large water content. He notes that the submerged diffuser is particularly useful in conjunction with burial pits, depressions, and dykes. Advantages of the submerged diffuser system include increased control over placement of both capping and contaminated sediments, decreased scouring of the bottom following impact, and less release of contaminated material into the water column.

There are disadvantages for all three methods for placing material to a uniform depth of say 10 cm or 20 cm because it is difficult to assure uniform placement. The drawbacks of using a submerged diffuser system is the requirement for a continuous source of material. In this study, it is assumed that the diffuser design would be most appropriate because of its ability to place material more uniformly and more exactly. This design is not particularly needed to minimize the impact of releasing contaminants during the operation because it is assumed that the clean fill is not contaminated. Hence, the logistics of placement will essentially require the following steps:

1. Digging up of clean fill from some other aqueous site or from land-based site;
2. Transporting this material to a supply barge.
3. Moving the supply barge out to a barge containing a submerged diffuser system which is continuously (semi-continuously as barges are connected and disconnected) placing material.
4. Continuous working of the supply barge during the day; its motion would be controlled with motors from the rear and from the sides to provide for control of its position.
5. Use of a land-based guidance system with appropriate communication devices and an on-board system which would control the barge's position to plus or minus 1 metre.

With respect to source of material, volume estimates would indicate that approximately $13.6 \times 10^6 m^3$ of clean fill would be required to bury all Upper Bay of Quinte sediments with fresh fill with a thickness of 10 cm. Such a thickness could be thicker or thinner; however, present data do not allow one to determine the optimal thickness. Accordingly, it is assumed that a 10 cm thick burial zone is used. If this option were adopted in the future, a more detailed analysis of the actual thickness of fill should be done.

For such a mass of material, approximately 136 ha of land would have to be extracted to a depth of 10 m to provide this volume of fill. In other aerial units, this is equivalent to about 335 acres (1/2 square mile) which is equivalent to extracting one-half of a one mile by one mile rural concession to a depth of approximately 35 ft. Such an extraction would pose substantive impacts to the landscape in the area to be extracted and would require a detailed environmental assessment. Alternatively, if a shallower depth were extracted, such as 3 m (10 ft.) of depth, then the area required would triple to approximately 400 hectares (1.5 square miles). This would cause an even larger scar on the landscape which would need to be carefully landscaped and revegetated.

An alternative source of material is to dredge relatively clean beach sand from the near-shore area of Lake of Ontario adjacent to the Bay of Quinte. We have not, in this report, assessed any particular sites that could be dredged, but typical data for such sands in sediments in 20-30 m depths of water characteristically have phosphorus contents at low enough levels that it would be a relatively good source material for "clean fill". The major question with respect to this material is whether other contaminants are present in the sediments which would be released to contaminate the water column of the lake during dredging of material or the Upper Bay of Quinte during the placement of the material. Also, the relative amount of contamination in the new sediments compared to existing sediments in the Upper Bay would have to be quantified and assessed. At present, due to the lack of sufficient data characterizing the level of contamination in the present sediments of the Bay of Quinte (heavy metals, toxic organics), it is not possible in this report, to assess the relative degrees of contamination. Accordingly, it is assumed that the degree of contamination is minimal for the purposes of study but such work will need to be conducted.

The logistics of placement thus involve the following: position the dredge at the proper depth in Lake Ontario, dredge the material from the bottom of Lake Ontario, with a

stationary barge, transfer it to the supply barges, transport the clean fill to the Upper Bay of Quinte, transfer the transported clean fill to the placement barge, and deposit the clean fill.

Estimates of the time for placing the clean fill are also required. Assuming that the placing barge would have a submerged diffuser of the order of 5 m in width, then approximately 27,200 km of transit distance would be required back and forth across the Bay in passes for the sediments of the Upper Bay of Quinte to be buried. Assuming a barge speed of 8 km/hr. during placement operations and an active working day of 6 hr./day, (which translates into a net barge speed of 48 km/day), then approximately 570 barge days would be required for placing. If 6 barges were working continuously, this could be carried out in a 95 day period. If only 5 barges were working, then 114 days would be required. If, only one barge could work due to rental considerations and/or supply considerations, and if the working period were 180 days per year, approximately 3 years of activity would be required to complete the dredging operation. For purposes of this study, it is assumed that 3 sets of barges would be working such that sediment burial could be completed within one ice-free season, i.e., 180+ days.

This would necessitate, an equivalent number of barges which are sitting in position on Lake of Ontario dredging clean fill and an associated fleet of supply barges for transporting the dredged material from the dredging site to the emplacing site in the Upper Bay of Quinte. Fewer or more than 3 dredging barges may be required, depending upon their capabilities. Assuming that a supply barge would be approximately 10 m x 20 m and be able to carry approximately 5 m high of dredged material, (hence, 1,000 m³) 24 barges are required as a supply fleet for each emplacing barge because each emplacing barge, moving 48 km/day having an application width of the submerged diffuser of 5 m and putting in .1 m of material, would require approximately 24,000 m³ per day of material.

Hence, the operation would consist of a specialty barge which is dredging out in Lake Ontario, an emplacing barge on the Upper Bay of Quinte, and a supply fleet of approximately 24 barges for supplying the emplacing barge and accepting material from the dredging barge. With their probable scope of being able to move at the rate of up to 8 km/day and working a 12 hour day, each barge would be able to move at the rate of about 100 km/day. Hence, given the logistics involved, it would appear that the distance

between the supply point and the emplacing point could be up to 80-100 km from the point of dredging the clean fill to the point of the emplacing barge, a distance which given inspection of the maps, would seem feasible.

3.8 Logistics of Implementation of Aeration

3.8.1 Enhanced Circulation

The choice of device for aeration will depend upon detailed numerical calculations, keeping in mind that the Upper Bay is not thermally stratified and that the primary objective is aeration of bottom sediments and not destratification in the Upper Bay. A design constraint will be to aerate a sediment microzone with minimal turbulence to avoid resuspension. Such calculations are beyond the scope of this report.

Compressors could be located on shore or on floating platforms while water pumps would be located on floating platforms. Electrical power lines would be required. Operation of the devices is labour-free after start-up but would require regular maintenance inspections. Aeration equipment is commercially available from suppliers in North America and Europe. Choice of compressors is discussed in more detail in Section 4.8.1.

Consideration must also be given to designing a diffuser system if air compressors are used. The diffuser system is essentially a system of plastic hoses distributed on top of sediments through which air is discharged. Water circulation is only enhanced at depths above the discharge port. Sophisticated numerical models will be needed to predict the optimal size and quantity of compressors and the location of air discharge ports in the Bay (see Section 4.8.1).

3.8.2 Logistics of Oxygen Injection

To estimate the number of oxygen storage barges, several assumptions are made. It is assumed that the WOC (water oxygen consumption rate) is similar to that of the hypolimnia of the Middle and Lower Bay (namely 50-100 $mg/m^3/d$). To account for the extra effect of algal respiration in the epilimnion, the WOC is assumed to be three times the volume O_2 depletion rate (VOD) giving a net rate of 0.200 mg/L/d. Such ratios are typical of Hamilton Harbour, which has similar levels of total phosphorus and chlorophyll a as the Bay of Quinte. If a design current velocity of 1 cm/s, which is rather low, is

assumed, a pocket of water would move approximately 4 km and lose 1 mg/L if there were no oxygen replenishment. If one desired to maintain a maximum DO depletion of 1 mg/L, the horizontal analyses would suggest that an aerator should be placed at approximately a 4 km interval. This would require ten such devices along the Bay.

Vertical stratification influences the ability of a water body to replenish lower water which is relatively non-oxygen rich with oxygen-rich, overlying water. However, in the shallow Upper Bay, transient stratification would be observed, especially in the deep holes (Z max = 15 m). In such areas, longer periods of time could occur without replenishment. Accordingly, this would suggest that it would be useful to increase the density of oxygen injection devices in such areas. For illustration purposes, it is assumed that ten aerators using liquid air would be placed on the Bay, concentrated in the areas of deeper water.

It is assumed that injection would be from barges would be anchored on the Bay and contain a storage tank. Conventional storage tanks are 30 feet high, 15 feet in diameter and contain approximately 6,000 tonnes of oxygen (supplied by Liquid Air). For purposes of stability on the lake, it is assumed that the tank would be placed horizontally in the barge.

The mass of oxygen required would supply a 4 km^2 area, average depth of 3.5 m with an oxygen demand of 0.2 mg O_2/L/day. The net requirement is 2.8 tonne/day per 4 km^2 area or 28 tonne/day for the ten aerators. This should be viewed as a conservative figure, because this assumes that all oxygen consumption in the water is replenished by oxygen injection, when in fact it is also replenished by natural processes such as algal photosynthesis and reaeration.

Due to aeration efficiencies, only part of the oxygen injected into the water column will be absorbed. With the design being tested by Environment Canada in Hamilton Harbour, an aeration efficiency of 80% is achieved in a 10 m deep water column (in testing in the southwest end) and 43% in a 4 m water column (based upon pilot scale). Because the average depth of the Upper Bay is 3.5 m, an average efficiency of 40% is assumed. This would, of course, be larger in areas where the water is deeper. For this value, the oxygen required would amount to 7 tonne/day per installation. Assuming an injection period of 4 months (mid May to mid September), the total air required would be 8,400 tonnes of oxygen.

3.9 Lake Drawdown

Lake draw down is infeasible.

4.0   COST OF IMPLEMENTATION

The annual operating and maintenance costs and initial capital costs have been estimated using available data.   Total costs are reported on an annual cost basis.   An appropriate discount rate was developed by examining borrowing costs.   Fixed mortgage rates from the banks for home mortgages are currently 11.5% for a five-year period, and 12% for a ten-year period.   For costing purposes of this section, it is assumed that capital borrowing costs are 12%.

The following headings are used.

4.1   Null Option (Natural Clean-up)
4.2   Chemical   4.2A   Alum Salts
                     4.2B   Ferric Salts
                     4.2C   Calcium
4.3   Biochemical Control
4.4   Fly Ash
4.5   Barriers
4.6   Dredging
4.7   Sediment Burial
4.8   Aeration
4.9   Lake Drawdown

4.1   Null Option

There are no direct costs associated with the null option.   Indirect costs such as associated with control of point or non-point sources are not involved here.   They are given in the report by Canviro (1987) and Ecologistics (1987).

4.2   Cost of Implementation of Chemical Control

4.2a   Alum

A summary of costing information available to 1986 is shown in Figure 4.2.1 (after Cooke et al., 1986).   The principal component of cost is labour.   The different lakes involved
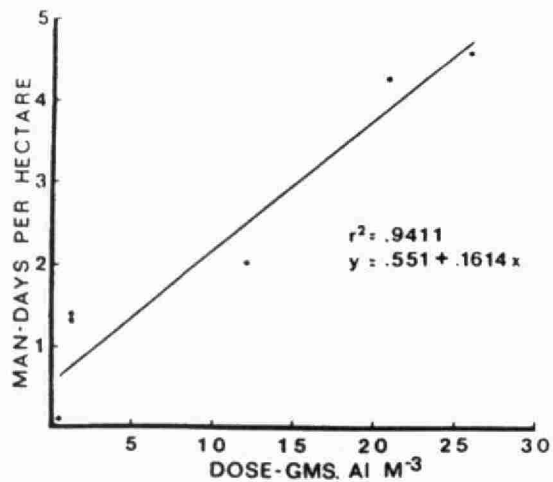
FIGURE 4.2.1:   Relationship between dose of aluminum sulfate and labor (man-days ha^{-1}) for six North American lakes (see text for lakes and data sources)

are: Horseshoe Lake, Wisconsin (Peterson et al, 1973), Welland Canal, New York (Shannon et al., 1974), Dollar Lake and West Twin Lake, Ohio (Cooke and Kennedy, 1981a), Medical Lake, Washington (Gasperino et al., 1980) and Liberty Lake, Washington (Cooke et al., 1986). As Cooke notes, the linear relationship ($r^2 = 0.941$) from these six data points provides a guideline from which one can estimate the major expense, labour, by obtaining the necessary data on alkalinity, pH and lake volume and calculating the required dose.

As of 1986, Cooke et. al., note that there is not a commercial apparatus for alum application. They also argue (1986) note, the cost of application could be lowered if a more rapid application technique could be developed. At the 1987 NAWMS meeting, a commercial vehicle was available which could hold up to 1400 gallons of liquid (a firm from Sweetwater). High velocity-high volume shore-based pumps and hoses should be tested.

As indicated in Section 2.2 and 3.2, the design considered in this report for applying alum is the modified weed harvester of Connor and Smith (1986). Cost data developed by Connor and Smith are given in Table 4.2.1. One lake in Table 4.2.1 is also included in the data of Figure 4.2.1. Their data suggest that the weed harvester is the cheapest approach.

For evaluating application to the Bay of Quinte, the individual components are costed assuming application by a weed harvester and the manpower estimates of Connor and Smith (1986) scaled up. Accordingly the data presented above are not reconciled.

The following costs are estimated:

1. The chemical costs are estimated at $10.1 million. This assumes that the aluminum dose is 10 mg Al/L and based upon a quoted delivery cost of $0.22 per kg for alum, delivered in 6,000 gal lots. Assuming the need to repeat the treatment every three years, the annual equivalent cost is $5.38 million/year.

2. Application costs for labour are estimated at $4,500,000. This is based upon 8,100 crew-days for application, a 10% allowance for extra crews to set land-based siting for the navigation guidance system on the boat and a per diem for the crew of

TABLE 4.2.1: Equipment, Labor and Chemical Cost Comparisons for Aluminum Salts Injection

Lake/Year	Area Treated (ha)	Aluminum Dose (g Al/M³)	Costs for Chemicals, and Labor*	ha Treated/h	Man day/ha	Cost/ha
Dollar Lake Ohio 1974	1.39	20.9 Aluminum Sulfate	$589 (Chemicals Only)	(0.67)**	4.30	No Data
Medical Lake Washington 1977	60	8.0 Aluminum Sulfate	$132,093	(0.67)**	No Data	$2,202
Annabessacook Lake Maine 1978	121	25 Aluminum Sulfate and Sodium Aluminate	$234,060	0.67	1.12	$1,934
Kezar Lake New Hampshire 1984	48	30 Aluminum Sulfate and Sodium Aluminate	$65,604	1.20	0.50	$1,367

* All cost estimates were corrected for inflation (Krefgtz, 1982).
* Assumed to be similar to that of Annabessacook.

$500/day. Based upon an application of every three years, the annual equivalent cost is $1.85 million/year.

3. Cost of the rental of the application barges is estimated as $1.78 million based upon 1782 weeks of rental at the rate of $1,000/week. It is assumed that the required navigational system costs are included in those of the barge. This is equivalent to an annual cost of $742,000/year.

4. Fuel costs are estimated as $200,000 based on the analysis given for sediment burial.

The total cost is thus $8.0 million/year. On a per treatment basis, the total cost per treatment is $1,400 per hectare which is comparable to the costs of Connor and Smith (1986).

4.2b   Ferric Chloride

The costing of ferric chloride is similar to that of aluminum with the exception of the chemical costs. The following are the estimated costs.

1. The chemical dose is estimated as $7.31 million per treatment. This assumes an application rate of 4 mg Fe/L (as per Foy, 1985; and Hoestra and Marwald, 1985) and a supplied cost of $1.32 per kg for $FeCl_3$ delivered to the site in 6,000 gal lots. The frequency of treatment is not known. Assuming a three-year application interval, the annual cost is equivalent to $3.0 million.

2. The application costs are assumed to be equivalent to those for alum. They are:

a)   labour: $1.85 million/year
b)   barges: $0.742 million/year
c)   fuel: $.08 million/year

The total annual cost is $5.6 million/year.

4.2c   Calcium Carbonate

The costing for calcium carbonate is as follows:

1.   The chemical cost is $670,000 per treatment. This is based upon the treatment dosage used by Prepas and Murphy (1987) of 14 mg/L $CaCO_3$ plus 13.2 mg/L $Ca(OH)_2$ to Figure 8 Lake and a delivered cost of $38/tonne to the Belleville area in bulk truck (dry form). Assuming an annual application since the desired frequency is unclear, the estimated annual equivalent cost is also $670,000/year.

2.   Capital costs are estimated as $100,000. This is to provide for shore-storage facilities for delivered calcium carbonate and a delivery system to the barge. Assuming replacement every five years due to wear, tear, and corrosion, the annual equivalent cost is $27,700/year.

3.   Costs for logistics of application are assumed to be similar to those for alum. Due to the annual application, the costs are:

a)   labour $4.5 million/year
b)   equipment $1.78 million/year
c)   fuel $0.08 million/year

Accordingly, the application costs are estimated as $7 million/year

4.3   Cost of Implementation of Biochemical Oxidation

The two methods for biochemical oxidation: the RIPLOX method and addition of solid calcium nitrate are costed in this section.

The cost of treating a system with the RIPLOX method is approximately $10,000 per ha. (Atlas Copco, personal communication). Assuming that only the Upper Bay of Quinte is treated and that the treated area is 100 km^2, the cost of a single treatment is about $100 million. Even if the cost were reduced 50%, and the treated area were reduced 50%, the cost of a single treatment would likely be circa $25 million. Since multiple treatments would be necessary in the absence of additional point source and non-point source controls, this method is quite expensive for the Bay of Quinte. Assuming annual application, the annual operating costs are $100 million/year.

Calcium nitrate is produced by Norsk Hydro in Norway as a by-product in the manufacture of phosphate. It is distributed in Canada by W.G.M. Hydro Inc. of Toledo, OH (419-866-0777 attn: David Banachowski) at a cost of $140 per tonne FOB Toronto (or any Great Lakes Port). The total cost of purchasing and transporting 100,000 tonnes of calcium nitrate is approximately $15 million.

Application of calcium nitrate is assumed to be by modified weed harvesters (similar to alum). Assuming similar time requirements for application, the application costs are similar to those for calcite: $4.5 million/year for labour and $1.78 million/year for barge rental and $.08 million/year for fuel. Accordingly, the total annual cost for calcium nitrate is $20 million/year.

Recently, the Ministry of Northern Development and Mines examined the feasibility of mining a phosphate deposit near Hearst, Ontario in a report entitled "The Northern Ontario Phosphate Study" (Contact Geoff Minnes, 416-965-3128). The report has not been released yet but if the phosphate manufacturing process chosen produces calcium nitrate, then a large, local source of the material will be available at a substantially lower cost. A local market for calcium nitrate may make the phosphate mining venture more attractive, and is in keeping with Provincial and Federal job-creation initiatives for that region.

4.4 Cost of Fly Ash Application

The cost of fly ash application has two major components: transportation and application. As mentioned in Section 3.4, development of a technique suitable for application of 6 million tonnes of fly ash is beyond the scope of this report. Hence, only technologies currently available and applied previously to small lakes are considered here.

A fleet of 1500 MOVAB boats would be required to complete fly ash application within 1 month. At a cost of $30,000 each, the total fleet cost is $45 million. Assuming a 10 year period for amortizing the fleet, the annual equivalent cost is $8.0 million.

For the purposes of calculating transportation costs, it is assumed that 6 million tonnes of fly ash is available free at Great Lakes ports every year. Transportation by freighter

is assumed to be $15 per tonne for total annual transportation costs of $90 million. Manpower costs are based on 3 man-months per boat at $2,000 per man-month for total labour costs of $9 million.

Assuming that fly ash application were every year, the annual equivalent cost is $107 million/year. Assuming that the frequency of application is once every ten years similar to that assumed for the sediment burial option, the annual equivalent cost is $25 million/year.

4.5   Costs of Emplacing Barriers

The costing of a barrier is based upon the following points.

1.   The cost of material is estimated as $6.3 million. This is based upon the HRCA nylon costing $25,000 for 430 m length for one application (Christie Reservoir) in 1987 and 800' costing $17,000 two years ago for Valens Reservoir. In both cases, the curtain was approximately 2.5 m deep. Using 1987 figures gives a per unit cost of about $25/m^2. For the Bay of Quinte, approximately 250,000 m^2 of curtain would be required for the assumptions outlined in section 3.4.

2.   The manpower costs are estimated as being $220,000/year. This is based upon projecting the small lake application of HRCA. This then requires approximately an installation rate of 500 m of curtain per day by a three man crew (hence 145 crew days for installation), 290 crew days for removal (assuming 145 crew days to remove it, and an equal number to clean up the curtain), and a per diem cost of $500/day for the three-man crew.

3.   The boat costs are estimated as $58,000 based upon a weeky rental of $1,000/week and 290 days for installation and retrieval.

In terms of practicality, it is obvious approximately 10 crews would be needed to place the curtains in an acceptable period of time (15 days; 3 weeks). However, it is plausible that a barge or equivalent with a larger boom and other capabilities would be used to install the curtain at a faster rate. This would require both crews on shore to manipulate land-based transport and transfer of rolls of curtain to the water's edge and crews on

board to place the curtain, but would give a substantial increase in economy. This is not costed here in order to maintain a similar economic basis for costing of all control options.

These costs result in an annual cost of $1.5 million for capital, operating and maintenance in 1987 dollars. This assumes that the lifetime of the curtain is 10 years under normal wear and tear, and that 2% of the curtain must be replaced annually due to vandalism etc., and that capital borrowing costs would be 12%.

If vandalism required 10% replacement of the curtain annually, the annual capital operating and maintenance costs would be $2.0 million.

4.6   Costs of Dredging

The two main costs of dredging are the dredging operation itself and disposal. These costs are estimated as follows.

1.   The cost of dredging would be $136 million. This is based upon a dredging cost of $5 per cubic meter which is typically observed for dredging operations in Hamilton Harbour with a cutter head suction dredging operation. Some economy would result due to the scale of the Bay of Quinte but such economics are not evaluated herein in order to continue the use of costing data based upon small scale operations. For a life expectancy of 10 years over which it might be expected that point source and diffuse source control would become fully effective, the equivalent annual cost is $24 million/year.

2.   Disposal costs are estimated as $20 million. This is based upon requiring a disposal site of approximately 270 ha and 10 m deep. It is assumed the disposal site would be reclaimed lake area and that the land values, capital costs of placing the dredge spoil, capping and vegetating are $74,000/ha. These are typical costs developed by Hamilton Harbour and also represent land value that the Ministry of Transporation and Communications paid to the Hamilton Harbour Commission for the expansion of the Queen Elizabeth Way. This results in an annual equivalent cost of $3.6 million for disposal.

Accordingly, the net dredging costs are approximately $28 million per year.

By way of comparison, the dredging costs (excluding disposal) are equivalent to $10,000 per hectare whereas dredging costs in Windemere Basin of Hamilton Harbour, a highly contaminated arm of the Bay, 40 ha in size is estimated as $110,000/ha. The costs for this basin are higher due to the larger volume per unit area to be dredged.

4.7   Costs of Implementation of Sediment Burial

Two estimates of the costs of implementation of the sediment burial technique are made in this work.   One is an overall estimate based upon dredging data from small-scale operations. A second is a detailed analysis of the type of activities required for the scale of the Bay of Quinte.

For small scale operations, the conventional figure is approximately $5.00 m^3 of material dredged which would include dredging costs, removal costs, transport costs and disposal costs. Using this figure as a ball park value, the net cost would be $58 million. This thus assumes some economy of scale because the figure of $5/m^3 did not include long distances of transport from Lake Ontario to the Upper Bay of Quinte nor precise techniques required for emplacing the clean fill uniformly on the sediments to a depth of 10 cm across the Bay of Quinte.

A second way of estimating costs is to break down the logistics of operations into various components and to make estimates of these costs. The following estimates have been developed.

1.   The rental of barges would be estimated as $15.9 million for the complete operation of dredging clean fill, transporting fill through the Upper Bay of Quinte, and emplacing the clean fill in the Upper Bay of Quinte. This is based upon assuming that two specialty barges, one involved in placing the fill and the second involved in dredging the clean fill would be required for a period of 114 weeks (570 days, which assuming a five-day week become 114 weeks. Note, as explained in Section 3.7, this requirement of 114 weeks would be conducted by three sets of barges, each for 38 weeks.). The cost of the rental of these two barges would be $2,280,000. Supply barges are assumed to be required for 114 weeks at the rate of $5,000/week. For 24 barges, the total rental charge is $13.7 million.

2.   With respect to labour, the estimated costs are $5.6 million. This assumes that there are 2 workers per supply barge working a 12 hour day for 570 days at a rate of $15 per hour for 24 barges; this would cost $4.9 million. It assumes that there would be 4 workers on each of 2 specialty barges working 8 hours per day for 570 days paid at the rate of $15 per hour for a net charge of $540,000. Finally, it assumes that management personnel would work 570 days, 8 hours per day at a rate of $25 per hour for a total cost of approximately $114,000.

3.   Fuel charges are estimated as being approximately $1.2 million. This is based upon assuming that the specialty barges have a yield of .05 km/L consumed for the period.   For the barge involved in applying the dredged material, approximately 27,200 km would be transversed. Assuming a fuel cost of $.3/L, the total fuel cost for the emplacing barge would be $163,000. The total fuel bill for the dredging barge would approximately $55,000 based upon similar yields, assuming that most of its fuel usage is due to working on station, and that it consumes 30 litres per hour during the dredging operation. With respect to the supply barges, a fuel economy of .4 km/L is assumed, a transport rate of 100 km/day, 570 days of operation and a fuel cost of $.30/L. For 24 barges, the total cost is approximately $1 million for the supply barges.

Hence, based on a detailed analysis, the net charge for the specialty barges, labour, emplacement barges and fuel would be approximately $23 million. Assuming that the burial operation would have to occur once every ten years, then the net annual cost of this operation would be $4.0 million per annum. This compares to the cost estimate based strictly on the small scale cost estimate of $5/m^3 moved of $12 million per year. There is a difference of approximately three times between the two cost estimates. This then provides some indication of the possible sensitivity of using small scale cost estimates for estimate costs for a large scale operation. The difference could also indicate that some of the estimates used in the detailed approach are low; for example, the estimated rental costs for the barges at $10,000/week for the application barges and $5,000/week for the supply barges may be low and the time estimates may also be low. If that were the case, application of a 100% safety factor would then give the estimate for a total cost of $34 million once every ten years and hence a total estimated annual cost of approximately $6.0 million per year.

4.8   Cost of Implementation of Aeration

4.8.1   Enhanced Circulation

The optimum design and, therefore, cost, of enhanced circulation for the Upper Bay will depend upon detailed calculations which is beyond the scope of this report. However, a "ballpark" estimate is calculated for comparison of alternative methods discussed in this report. To arrive at an estimate, it is assumed that air compressors are used and that 30 SCFM of air flow per 10^6 square feet of surface area (9.3 ha) is required to achieve good mixing (Lorenzen and Fast, 1977). Hence, 408,000 SCFM are required in total for the Upper Bay.

Compair Canada Inc. distributes a variety of air compressors which may be suitable. Their largest high pressure compressor is a 300 H.P. P.S. rotary screw compressor rated at 1500 SCFM and 100 PSI. The cost is $59,000 + 12% + 7%. The total capital cost for 272 units is $19 million.

A low pressure blower may be suitable for use because the discharge ports will likely be in water less than 3 m deep. Atlas Copco distributes a centrifugal compressor rated at 100,000 SCFM and 20 PSI. The cost of 4 units is $2.5 million. A 100 SCFM centrifugal compressor is built by Joy Manufacturing of Kitchener (519-378-6400) attn: Mr. Hobler) at a cost of approximately $500,000. The total capital outlay for the Joy compressor would be approximately $2 million. In all cases, the cost of the diffuser system, transporation, installation and maintenance must be added to the above totals.

Choice of compressors is limited to rotary screw and centrifugal types. Above a certain H.P., other types of compressors are required by law to have a trained operator on site.

Finally, the power requirements for 408,000 SCFM is enormous which may require building a transformer station. As a technical representative for a compressor manufacturer stated, "The lights in Belleville will dim when the compressors start up."

The annual operating costs (including power, pipes and air diffusers) are estimated as $72 million per year and are based upon the following. Lorenzen and Fast (1977) estimate an average operating cost (including pipes and air diffusers) of $113,000 (U.S. dollars,

adjusted to 1983 conditions) for two air compressors producing air at 1200 cfm (34.3 cmm) at standard conditions. This value was adjusted for the larger volume of air required, 1987 cost conditions (using the composite ENR Index), and 1.3 Canadian dollars per American dollar.

The total equivalent annual cost, then, is $75 million. This is composed of $72 million for operating costs and $3.4 million for amortization of $19 million with a replacement period of once per 10 years.

4.8.2   Costs of Oxygen Injection

1.   The cost of oxygen is $462,000 based upon 8,400 tonnes and $55/tonne. The cost of $55/tonne is that being paid in Environment Canada's test in Hamilton Harbour.

2.   Equipment for injection and storing the oxygen is estimated a $300,000/year. This is based upon Environment Canada's cost of approximately for $7,000 for an installation which includes tank rental, moving charges, and diffuser and related engineering services for a two month period. This cost was doubled to take care of additional contingencies required for the Bay of Quinte.

3.   Additional management and labour costs are estimated as $41,000, based upon management personnel (1 day/week; $600/day; 17 weeks) and two maintenance personnel ($300/day each; 3 days/week; 17 weeks).

4.   Barge rental costs are estimated as $170,000, based upon $1,000/week per barge.

5.   Transport costs of the liquid oxygen are not known. Assuming production in Hamilton, at Liquid Air, a cost of $100,000/yr are assumed.

These costs total approximately $1.1 million/year.

Data for comparing these oxygenation cost estimates with those of traditional air systems, have been summarized by Cooke et al. (1986) as follows.

In one lake in which $32 \times 10^6 m^3$ of hypolimnion was aerated with 15 units in 1980 in a 420 ha lake, delivering 4.5 tonnes of oxygen per day, the total cost of the system was $2.1 million in 1983 dollars. This corresponds to $2.40 per km^3 of oxygen or $5,000/ha for 6 months of operation per year. In another installation, the average installed cost for six projects was estimated to be $354 plus or minus $216 per kilogram of oxygen per day. Power costs of 7 U.S. cents per kilowatt-hour resulted in an average operating cost of $0.056 per Kg O_2.

Hence, in comparison, the cost figures of the first operation when scaled from 4.5 tonnes per day to 70 tonnes per day would cost approximately $20 million U.S. (1983 dollars). Of concern here is the fact that these figures are based on 420 ha small scale operation as opposed to a large scale operation which has been costed out in the operation. For the costing data based on 6 projects, the average annual cost given would be approximately $4.9 million per year for the oxygen requirements of the Upper Bay of Quinte of 70 tonnes per day. This is composed of an annual operating cost of $470,000 primarily for power requirements, and annual costs of approximately $4.4 million required to amortize a capital cost of $25 million over a ten year period at an interest cost of 12%. Accordingly, the cost estimates for direct oxygenation are one-half to one order of magnitude lower than those from field data which employ aeration or enhanced circulation. The prime cost differences are the power differences and the number of units in the case of enhanced circulation.

4.9   Lake Drawdown

Lake drawdown is not costed because it is infeasible.

5.0 DETRIMENTAL EFFECTS OF CONTROL OPTIONS

The detrimental effects of the different control options are described in the literature on lake rehabilitation (e.g., Peterson et al., 1973; Dunst et al., 1974; Garrison and Knauer, 1983) in varying degrees of detail. These are critiqued under the following headings:

5.1	Null Option (Natural Clean-up)		
5.2	Chemical	5.2A	Alum Salts
		5.2B	Ferric Salts
		5.2C	Calcium
5.3	Biochemical Control		
5.4	Fly Ash		
5.5	Barriers		
5.6	Dredging		
5.7	Sediment Burial		
5.8	Aeration		
5.9	Lake Drawdown		

For a few control options, other ecological benefits are foreseen as plausible. They are also described in this section.

5.1   Null Option-Natural Cleanup

Internal loading of phosphorus to the Upper Bay of Quinte, via sediment release, will not be affected directly by this option. Therefore, no changes in the bottom fauna or fish would be expected, and the lake would continue to harbour a stressed community typical of eutrophic conditions until total phosphorus loading decreases.

Internal loading could be indirectly affected by control of point source and non-point sources. The mechanisms involved are described in Section 2.0. With reduced levels of oxygen consumption, the benthic community could change to a broader diversity of organisms. Such changes would generally not be viewed as a detrimental effect, but rather would represent an environmental benefit.

5.2   Detrimental Effects of Chemical Addition

In evaluation of the detrimental effects of chemical control, the literature generally characterizes the technique, under the heading of phosphorus inactivation/precipitation, as creating potentially toxic conditions (eg., Cooke, 1983). However, this terminology is directed towards the effects of aluminum because the application of other compounds such as iron and calcium have not been considered in such descriptions.

In this section, the detrimental effects of each technique are critiqued. However, due to the potential toxicity of aluminum, more attention is given to a description of its detrimental effects. Beneficial effects are also observed. For example, use of calcium, alum, or iron to prevent the release of phosphorus from sediments will improve water clarity if reflected in a change in algal abundance.

5.2a   Detrimental Effects of Application of Alum

The potential for aluminum toxicity to fauna has been demonstrated by Craig et al. (1985) in their development of a criteria document for aluminum for the Ministry of the Environment. The summary of the toxicity of aluminum is given in Figure 5.1.1. In acidic pH ranges, it is plausible that the aluminum ion itself is the toxic agent. In the neutral range, it is possible that formation of polymers on fish gills is the cause of toxicity. If aluminum polymers form on fish tissue, they affect respiration and transfer of other constituents across the gills.

The detrimental effects of aluminum addition to lakes has recently received some attention. One of the more comprehensive studies is the alum application to Bull Head Lake (Narf, 1985). In this lake, observations on residual aluminum levels and the impacts on the aquatic ecosystem were followed for a period of 4 years after application. Narf found that the species of green algae and diatoms increased and the number of blue-greens decreased after addition over the summer time. Zooplankton also increased in density. Rotifer abundance increased dramatically and there was an increased number of benthic species. Accordingly, it was concluded that addition of aluminum hydroxide had little or no toxic effect on the aquatic ecosystem. However, the impact of aluminum on the fisheries of the lake was not evaluated.

FIGURE 5.5.1: Zones of Aluminum Tolerance Over a Range of pH Conditions for Various Species & Life Stages
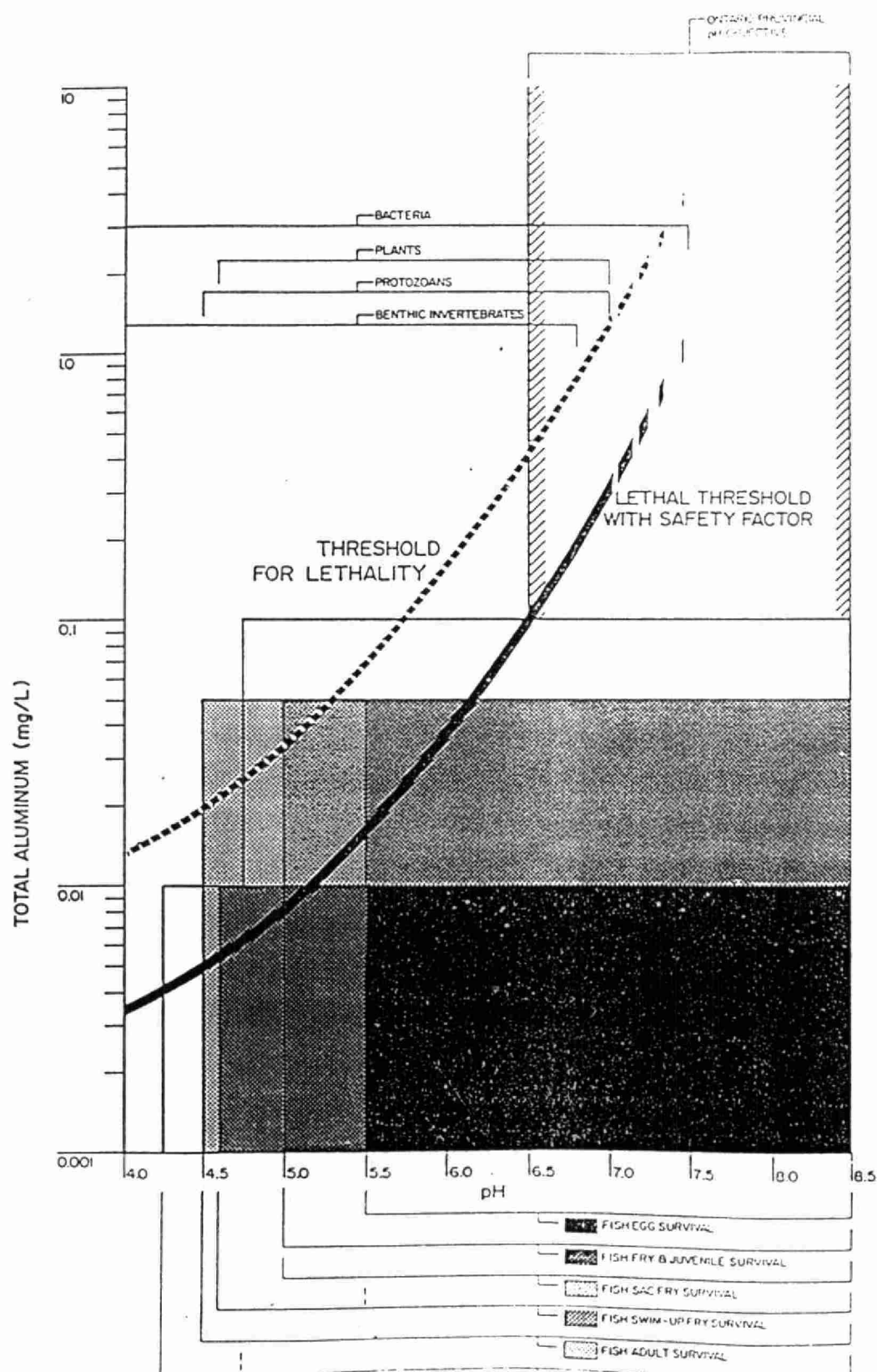
The best potential guide for assessing the detrimental impacts of aluminum would be to inspect observed residual concentrations of aluminum given in the literature and compare it to the toxicity diagrams prepared by Craig et al. (1985). Alternatively, the approach advocated by Kennedy and Cook (1982) could be adopted. They note that Freeman and Everhart (1971) using concentration bioassays, found that dissolved concentrations of aluminum below 50 ug/L of aluminum had no adverse effect on rainbow trout and that Peterson et al. (1974-76) obtained similar results for salmon. They suggest that a target of 50 ug/L of aluminum is a safe upper limit for post-treatment, dissolved aluminum concentrations. They conclude that a 50 ug/L level of aluminum left in solution is a useful criteria for examining the impacts of aluminum on the ecosystem.

Analysis of solubility considerations in the pH range of 5-9 shows that, irrespective of dose, residual concentrations should be well below 50 ug/L of total aluminum. Accordingly, application of alum in this pH range should result in a low enough residual level of total aluminum.

The other major potential impact of aluminum addition is the formation of aluminum hydroxide polymers which remain in solution and hence, which raise the total aluminum level above that suggested by solubility. This is a potential danger particularly if polyaluminum chloride (PAC) is added rather than alum. PAC has a polymeric structure which would be measured as a part of the dissolved phase and provide for aluminum concentrations above that set by solubility. Kennedy and Cooke (1982) suggest that one way to avoid leaving significant residual concentrations of aluminum hydroxide polymers is to make sure that sufficient aluminum is added that aluminum hydroxide precipitation occurs rapidly and that large-sized flocs are achieved. Formation of large-sized flocs would leave low residual concentrations of aluminum in the water column because large flocs settle rapidly to the sediments.

Indirect impacts of aluminum on fish have been described by Welch et al. (1986) in their addition to 2 m deep Long Lake. They note that the main changes were in the structure of the fish community. The changes generally were attributed to increased predation and growth by piscivores and changes in the numbers of planktivores due to reductions in phytoplankton concentration as a result of reduced internal phosphorus loading. There was no obvious toxicity effect.

Welch et al. (1986) also noted some impacts of alum addition during the post treatment phase on macrophyte densities and plankton community structure. The effects were complicated by the drawdown of the lake which occurred one year prior to alum addition.

Presently, aluminum is not considered to be a chemical of any health significance in drinking water (Craig et al., 1985). However, there have been recent press reports of epedemiological studies correlating aluminum concentrations in surface waters with incidence of Alzheimer's disease (see also Crapper et al., 1973). The United Kingdom has set a maximum acceptable limit for drinking water supplies of 200 ug/L, with a preferable concentration of 50, if possible (Gardiner and Manee, 1984). Such levels should be readily achieved if alum is applied to lakes with a pH range of 6 to 9; such levels are typically found in water treatment plants using alum for coagulation.

5.2b   Detrimental Effects of Application of Iron Salts

Unlike phosphorus inactivation by aluminum, which is generally noted to be potentially toxic (e.g., Cooke, 1983), application of iron salts are not specifically noted to be toxic. The main effects are:

1.   Higher iron concentrations in sediments may change the bacterial community in the sediments.
2.   Additional iron may diffuse into the water column, and influence the operation of water treatment plants (this is not forseen as being significant when the Upper Bay of Quinte is generally aerobic and well mixed).

With respect to the effectiveness of iron addition for controlling internal phosphorus loading, the literature provides some guidance. However, the assessment is not straightforward because iron applications have usually been to lakes which continue to have anaerobic hypolimnia. Additional work would be required to assess its potential effectiveness in the Bay of Quinte and any changes in biological cycles which may result.

5.2c   Detrimental Effects of Application of Calcium Salts

The application of calcium salts for control of winterkill in hard water lakes has been justified (Prepas and Murphy, 1987) because it has no significant detrimental effects upon

the lakes. Its use is being promoted as a substitute for copper sulphate which has traditionally been used for algae control in the prairie ponds, sloughs and small lakes.

Dramatic changes in bottom biota and fish are noted when lime is applied to acid lakes; however, fewer changes have been documented when applied to hard water lakes. The application of calcium carbonate to Figure 8 Lake was followed by a dramatic decrease in chlorophyll a from 200 ug/L to 5 ug/L over two weeks and a sharp increase in Secchi disk transparency. However, the mechanism involved in this decline is not obvious. An equivalent of 14 mg/L $CaCO_3$ was first applied to Figure 8 lake. The total calcium in the water remained constant in the succeeding two weeks, alkalinity increased by 20% and pH declined from approximately 9.5 to about 8.5, and total phosphorus and soluble phosphorus remained about constant during this period. After a subsequent application of calcium hydroxide (equivalent to 13.2 mg/L as $CaCO_3$), when the chlorophyll a levels had increased to 200 ug/L, chlorophyll a levels did not substantially decrease, calcium and alkalinity declined, total phosphorus remained about constant, and Secchi disk remained about constant. Most of the chemical effects appear to have been caused by changes in biological cycles.

The main effects noted in Figure 8 Lake over the next several months after treatment were two-fold.

1.   Despite the calcium not staying in solution immediately after its application, it appeared to return to the water column over the winter period.
2.   Internal loading of phosphorus was lower in the winter following application than in previous winters.

The data allow one to formulate interpretation of the degree of effectiveness, but such analysis was not done by Prepas and Murphy (1987) and would require additional work in order to provide an interpretation of its effectiveness to the Bay of Quinte.

5.3   Detrimental Effects of Biochemical Oxidation

Possible detrimental effects of the biochemical oxidation techniques are nitrate and nitrite toxicity and the physical impact of the RIPLOX method on benthic fauna caused by 'raking' sediments with the nitrate injector. However, the latter may be negligible

because sediments which are anaerobic, even if only for part of the year, are not likely to harbor a rich benthic community. Nitrite is considered here because it is an intermediate compound in the nitrification and denitrification processes. However, significant accumulation of nitrite is unlikely (Foy, 1986) because nitrite is unstable.

Localized nitrate levels will depend upon: 1) the time of year because the rate of denitrification is temperature-dependent; and 2) the choice of method. Nitrate concentrations will gradually decline following a discrete addition, due to denitrification in the sediments and hydraulic washout. Concentrations in the Bay may be more constant in the case of STP nitrification. Higher concentrations would be sustained in the vicinity of STP discharge pipes which is probably a high impact zone in any case. High concentrations will also occur at the sediment/water interface of 'raked' sediments injected with calcium nitrate.

The maximum acceptable level in drinking water (Health and Welfare Canada 1978) for $NO_3 + NO_2$ is 10 mg N/L. The maximum acceptable level set by Ontario (OMOE 1978) is 10 mg NO_3-N/L and for NO_2 the maximum acceptable level is 1 mg NO_2-N/L.

The Ontario (OMOE 1978) water quality objective for $NO_3 + NO_2$ in livestock water supply is a maximum of 100 mg N/L. The objective for NO_2 in livestock water supply is a maximum of 10 mg NO_2-N/L.

Toxicity of nitrate and nitrite to aquatic fauna has been reviewed by Colt and Armstrong (1981). The 96-h LC50 value of nitrate to aquatic animals ranges between 1000-3000 mg N/L. In most cases, the nitrate was added as $NaNO_3$; accordingly the toxicity results may be complicated by high levels of Na^+. Gradba et al., (1974) found that 5-6 mg N/L as NO_3 caused significant increases of ferrihemoglobin in the blood of rainbow trout, serious damage in the peripheral blood and hematopoietic centres, and serious damage to liver tissue. There is a possibility that nitrite was produced in culture water by bacteria and that toxicity was caused by nitrite. Nitrate levels of 90 mg N/L had no effect on the growth of channel catfish and levels of 200 mg N/L had no effect on Peneid shrimp.

The 96-h LC50 value of nitrite to fish is variable depending on water chemistry and species. The ranges reported in the literature are 12.8-13.1 mg N/L for channel catfish, 0.20-0.40 for rainbow trout, and greater than 61.0 mg N/l for the mottled sculpin.

Chronic exposure of steelhead trout to 0.03-0.06 mg N/L for six months in soft water caused no significant growth reduction, changes in gill histology, blood chemistry or impaired ability of the smolts to adapt to seawater.

Assuming a daily dosage of 76 tonnes of NO_3-N to the Upper Bay is instantaneously mixed and consumed completely each day (some will be transported downstream), then the average NO_3-N concentration upon instantaneous addition to the Bay will be 0.16 mg/L, which is well below Federal and Provincial guidelines for drinking water. Toxic effects on aquatic biota are also unlikely, based on reported 96-hour LC50 values which are above the 1 mg/L range. However, the study by Gradba et al. (1974) with rainbow trout should be reported.

There is a risk of increasing P loading if the calcium nitrate utilized is a by-product of phosphate manufacturing. Assuming that the phosphate impurity is 1%, then the additional P loading is 1300 kg/day, which is equivalent to the daily internal P load prior to control (Minns et al., 1986). To be feasible, the phosphate content should be less than 0.01%.

5.4 Detrimental Impact of Fly Ash

There are three areas of concern to consider with respect to the effects of fly ash application:

- o heavy metal content
- o physical impact on biota
- o maintaining navigability of the shallow Upper Bay.

Fly ash contains variable quantities of heavy metals which are contained within the fly ash matrix and adsorbed to it; these metals may enter the food chain via accumulation in benthic organisms. Chemical analyses to quantify relative increases in sediment metals concentrations will be needed. Bioassays to quantify body burden increases will also be necessary. Chemical elutriate tests to estimate the relative rate of release of contaminants (heavy metals) also would be required.

Fly ash application will temporarily increase suspended solids which may have a detrimental affect on macrophyte beds and gill breathers. The introduction of a layer of fly ash over the sediments may smother benthic organisms and affect fish spawning success. Burial of spawning grounds can be minimized if the locations of important spawning grounds are known and avoided.

Finally, adding large quantities of material to a shallow bay with a high sedimentation rate is not desirable because it may necessitate frequent dredging to maintain navigability.

5.5   Detrimental Effects of Placing Barriers

Both curtains and macrophyte beds are barriers which affect current velocities. Both are described in Section 2.5, but only curtains are assessed from the point of view of logistics (Section 3.5) and costs (Section 4.5). The detrimental effects of both are described here.Curtains and similar devices could be placed in the Bay of Quinte to reduce lake currents and sediment erosion of particulate phosphorus. It is difficult to assess its potential effectiveness unless the percent of internal loading from sediment erosion is known. Barriers could be placed only in critical local areas containing high phosphate concentration in the sediment.

A   Curtains and Booms

Dependent upon designs, the curtains would have a substantial to a very catastrophic effect upon the ecosystem of the Bay. It would also have a substantial social impact. Booms would have a small to large impact dependent upon their scale of installation. Furthermore, the actual effect upon internal loading is not known.

Assuming a design which involves extending curtains from near shore to the middle of the Bay but leaving a space of approximately 100 m in the middle for boat traffic, the following impacts could result:

1.   Sail boat activity would be severely affected;
2.   The curtains would not completely minimize internal loading because bottom currents along the middle channel would continue to erode sediments there, and also in areas cordened off by the curtains and adjacent to the channel.

3.   Sediments would be disturbed where the curtains are anchored to the sediments; this would increase turbidity in the water adjacent to these areas and result in impacts analogous to those of dredging.

4.   Bottom fauna and fish migration patterns would be disturbed.

<u>B</u>   <u>Macrophytes</u>

The detrimental effects of increasing macrophyte areas are forseen as being relatively smaller than those of the curtains. However, one cannot directly compare these two options as laid out in this report because the curtains are assumed to be placed completely across the Bay, whereas macrophytes would only grow in shallow nearshore areas.

The impacts of the macrophyte beds would include the following:

1.   The bottom sediments would be stabilized by the macrophytes and fishery habitat increased; existing bottom fauna and invertebrate populations would be displaced by the beds.

2.   Phosphorus recycling from the sediments may be increased if the macrophyte beds are dominated by species which obtain phosphorus from the sediments. However, erosion of particulate phosphorus into the water column would be decreased.

5.6   Dredging

Dredging may temporarily increase the rate of nutrient release from bottom sediments by agitation and suspension. This also applies to the release of toxic chemicals, such as heavy metals, pesticides and PCBs, which may find their way into the food chain. Dredging will also greatly reduce benthic invertebrate populations temporarily reducing the food supply temporarily for bottom-dwelling fish. Disturbance of the sediments may also affect on spawning success.

The literature for assessing the impacts of dredging upon the environment is vast; accordingly, in this work, a limited number of studies are used as a source for characterizing and critiquing the impact of dredging upon the environment. The study of the impact of dredging has centered upon two main areas:

o   the residual contaminants including suspended solids, metals and organics which are left in the water in the vicinity of the area being dredged,;

o   studies on the characteristics of the disposal site itself.

The two main types of dredging which have been assessed are the clam bucket and the suction dredge. Many studies have evaluated estuarine systems; several have evaluated lacustrine systems. In particular, the US Army Corps of Engineers has taken the lead in developing protocols for dredging, as a part of their mandate for keeping navigable channels open to shipping in the U.S. This report draws particularly on the results of these studies in making a generic assessment of the impact of dredging upon the lake water quality of the Bay of Quinte.

Impacts in Areas Being Dredged

Palermo et al. (1984) compared the effect of different dredging techniques upon contaminant release to the water column during the dredging operations. They found that the water-tight, clam-shell dredge substantively reduced the amount of suspended solids in the upper part of the water column compared to the conventional open clam dredge. Of the various dredges: dust pan head, the cutter head, and the hopper type, Raymond (1984) concluded that the hopper dredge was least desirable of all three if sediment resuspension proposed an environmental hazard. Furthermore, Raymond recommended that the cutter head dredge be used wherever possible to minimize resuspension. He also found that the dust pan head did not significantly reduce the amount of suspended solids in the water column in contrast to some of the literature which described a potential reduction of suspended solids in the water column where the dust pan head had been used.

In many instances, the impact of dredging activity on suspended solids in the water column has been monitored after cessation of dredging. These studies give generally consistent results, but occasionally a mixed picture results. In marine systems, a plume is often noted downstream of the dredging area. This plume is maintained because of the river velocity and the small particles found in the plume. These fine particles have a larger percentage of contaminants than the rest of the sediments. In relatively more quiescent lake systems, and even in areas which current speeds are substantial, monitoring of the dredged area shows that there is a low content of suspended solids

found in the water. For example, dredging of sediments with the Hopper dredge, in an embayment in Lake Erie showed that ambient conditions returned within about an hour after cessation of dredging (Alther, 1984). In this case, the current speeds were substantive being up to 70 cm/s. It is possible however, that even though ambient conditions returned to the area being dredged, that a sediment plume moved with the currents to cause an excessive concentration of suspended solids in other areas of the lake. Monitoring of these other areas was not described in any detail by Alther (1984).

A second major impact of dredging is the release of contaminants such as organics to the water column after the suspended solids materials have settled out. A mixed picture has resulted from field studies. In a riverine situation, Sodergren (1984) in examining organic chlorine residuals from the water column, found that large quantities of particulate materials, dissolved nutrients and manganese and copper were released to the water but that cadmium concentrations in the water remained essentially unchanged. In other studies, very few increased chemical concentrations of metals, organics, or nutrients are noted. These low levels of impacts must result from the lack of desorption of contaminants from the particulate fraction of the substances into the water column and the rapid settling of the particulate materials. In addition, where increased metal and organic levels are noted, it is quite plausible that the materials released and remaining in the water column are colloidal in nature (i.e., passing a 0.45 micron filter), and are not truly dissolved. Such colloidal characteristics are required to explain the elevated level of metals such as iron found in solution. For example, the kinetics of oxidation of iron from Fe (II) to Fe (III) are so rapid that essentially all of the iron should be present as iron hydroxide, a solid phase; the associated ionic concentrations which are at equilibrium with a solid phase are of the order of micrograms per litre and lower, whereas the measured concentrations are in the milligram or submilligram per litre range. The lack of adequate characterization of the colloidal state of these metals in solution potentially could explain the divergence of results obtained in various studies. In other cases, water in the area of dredging is relatively uncontaminated partly because the sediments being dredged or dumped are sandy in nature and hence contain fewer contaminants. Sandy sediments also settle rapidly in comparison to finer sediments (for example, van Dolah et al., 1984).

Burks and Engler (1978) reviewed published literature and the results of the Dredged Material Research Program of the U.S. Army Engineer Waterways Experiment Station at

Vicksburg, Mississippi. They concluded that open-water disposal can have a temporary impact upon the receiving aquatic environment if the dredged sediments contain elevated concentrations of chlorinated pesticides, PCBs or ammonia. Harmful concentrations of heavy metals can be released from sediments at certain combinations of pH and oxidation-reduction potential, but probably would not be released by most typical dredging or disposal operations. Chlorinated hydrocarbon pesticides, PCBs, oil and grease compounds, heavy metals, and phosphates are rapidly sorbed by suspended particulate material that undergoes resedimentation in the water column.

Releases of high concentrations of organic and inorganic contaminants from sediments during trenching may lead to increased bioaccumulation producing sublethal effects on growth and reproduction and thus a decrease in biological productivity of less tolerant organisms. Disturbance of the sediment may also result in oxygen depletion of overlying waters due to decomposition of resuspended oxygen-demanding substances, greater primary production due to nutrient release, and the creation of floating scum and debris.

The total concentrations of contaminants in the sediments have little relationship to the amount released to the water column or to their availability to biological organisms. Many studies have shown that much of the total concentration of heavy metals in sediment material is not readily solubilized, as it is associated with less reactive phases such as found in the mineral lattice of crystalline solids, strongly sorbed to particulate surfaces, and in organic materials (Gibbs, 1973; Walters and Wolery, 1974; Brannon et al., 1976). Similarly, organic compounds such as PCBs and organochlorine pesticides, generally have a low solubility and are tightly sorbed by clay particles (White and Mortland, 1970; Young et al., 1976).

Standard elutriate tests can be performed on representative sediment samples to assess the potential release of these constituents during dredging. The following types of results are obtained.

1. Investigations of a variety of sediments have shown that ammonia is consistently released from dredged sediments during the elutriate test (Brannon, 1978). Ammonia may be hazardous to aquatic life if alkaline pH conditions exist at the site where the non-toxic ammonium ion (NH_4-N) can be converted to toxic un-ionized ammonia (NH_3-N). Average total ammonia concentrations ranging from

0.11 to 0.34 mg/L have been reported near the outfall of various dredged spoil disposal operations. During these previous studies, total ammonia concentrations as high as 3.25 mg/L have been observed, which exceed water quality criteria for un-ionized ammonia. However, it was concluded that these occasional transient elevated concentrations should not exert any significant impact on water quality.

2. Orthophosphate has not exhibited consistent release patterns during the elutriate test or disposal operations (Brannon, 1978).

3. Chlorinated hydrocarbon pesticides and PCBs are rapidly sorbed from aqueous solutions (Burks and Engler, 1978). This behaviour is generally the case for chlorinated hydrocarbon pesticides during the elutriate test. However, releases of PCBs ranging from 1.3 to 6.9 times the concentration in the receiving waters have been found in some elutriate tests, and appeared to be related to low oil and grease content (Brannon, 1978).

4. Manganese is the only metal released in substantial quantities during elutriate tests and aquatic disposal (Brannon, 1978). Manganese, however, is generally not toxic, and is an essential micronutrient. The most stringent standards for manganese (0.05 mg/L) are based on aesthetic considerations for potable water. Tolerance values reported for freshwater life range from 1.5 to 1,000 mg/L (U.S. EPA, 1976). The slightly elevated manganese concentrations reported for disposal site waters shortly following disposal were well below the critical tolerance range.

5. Consistent release of trace metals other than manganese has not been observed during studies of the elutriate test of dredged sediments (Brannon, 1978). Transitory releases (duration of a few minutes) of mercury (0.01 to 0.05 ug/L), lead (less than 0.04 mg/L), cadmium (0.00008 to 0.0025 mg/L) and nickel (0.005 to 0.020 mg/L) have been observed on occasion during dredge spoil disposal. Iron is usually released initially in much higher concentrations than metals other than manganese. However, released iron is subject to very rapid oxidation and precipitation in the water column. As discussed previously, the precipitation of iron oxides will tend to remove other metals and orthophosphate from solution by coprecipitation and sorption processes. Large releases of zinc have been observed during elutriate tests under anoxic conditions, but not during aerated elutriate tests. Aerated conditions generally prevail during dredging and disposal operations.

However, in assessing chemical release during dredging and disposal and its potential impact on water quality and bioaccumulation, mixing and dilution processes should be taken into account (Brannon, 1978).

Furthermore, a number of studies of contaminant concentrations in sediment cores from undisturbed sedimentary environments have revealed that a vertical concentration gradient may occur, with the higher contaminant concentrations occurring in the surficial sediments followed by a rapid decline to low background concentrations (e.g., Kemp et al., 1976). This surface accumulation of contaminants in the most recent sediments is the result of anthropogenic activities. The low contaminant concentrations in deeper sediments, on the other hand, represent precultural background levels due to natural geological and biological processes.

It is expected that, during the dredging operations, the mixing of contaminated surficial sediments with the deeper uncontaminated sediments will result in rapid and substantial resorption of contaminants released to the water column by the uncontaminated suspended sediments.

In addition to monitoring water and sediment areas for assessing impacts of the disposal of dredged spoils, various investigators have used synthetic systems to investigate impacts. Many of the studies were carried out or facilitated by the U.S. Army Corps of Engineers Waterways Experimental Station (for example Zaroogian, 1985; and Francingues et al., 1985). In laboratory systems, the mean oxygen concentration in flow water flowing from over the dredged material was ten times lower than that of inflowing water while the ammonia content of the outflow water increased substantively. This indicates that after disposal of dredged material, the sediment rapidly re-established its biochemical characteristics, involving such reactions as oxygen consumption and ammonia generation through hydrolysis. Accordingly, in addition to mechanical changes in the sediment structure as a result of disposal, biochemical profiles are rapidly established. This will affect the formation of dissolved substances in the porewater and the potential transport of this material upward to overlying water.

Direct impacts upon sediment and other biota and upon residual levels of nutrients and other contaminants have been noted after dredging. In a Swedish lake, Sodergren (1984) found there is no significant biological uptake of chlorinated hydrocarbons in fish

sampled in the area of a dredging operation but that the PCB content of surficial sediments actually increased ten years after completion of dredging. It is plausible that the lack of impact of dredging upon fish results from the fact that fish are migratory species which inhabitat a dredged area only for a short period of time compared to the total life cycle. The increase in sediment PCB content must have resulted from increased loadings to the lake (such as diffuse source runoff, inputs, or point source inputs). Such changes over time show that dredging must be viewed as strictly an interim control measure which is constantly required if loadings to the lake are not prevented.

Vertical migration of nutrients may be of potential concern even if sediments are dredged to a certain level. For example, Hart et al., (1986) concluded that the benthic community structure and bio-accumulation of contaminants would continue unless complete removal of deep contaminated sediments was implemented in the case of an assessment of radionuclide concentrations in Port Hope Harbour. Accordingly, the sediment profile of phosphorus with depth, and secondly phosphorus concentrations in flow waters are essential pieces of information for the potential impact of dredging and its long term effects on a biotic community and the sediments. Herodek and Istvanovics (1986) used the vertical distribution of phosphorus in the sediments to determine the extent of sediment removal which would be required to reduce the degree of internal loading. Such depth information, as was pointed out above, is not available for the Bay of Quinte. Accordingly, only a limited assessment of the potential of deeper sediment phosphorus can be made in this study.

Another study which examined the impacts of dredging on the benthic community was carried out by Ismail (1985). He found that there was a decrease in the density of benthic organisms, the number of species and the number of individuals after hydraulic dredging in the Delaware Bay area; however, the community recovered to pre-dredging levels within a period of 3-10 months. Dredging may also affect the standing crop of phytoplankton. In an assessment of the South Carolina estuary (van Dolah et al., 1984), and of a Quebec estuary (Lalancette, 1984), a decrease in standing crop of phytoplankton was noted. In the Quebec study, the decreased phytoplankton levels were attributed to an increase in turbidity which decreased rapidly and the impact was considered minimal (in both studies).

Impacts in Disposal Areas

A mixed picture has evolved in the literature of the type of contaminants released from dredged materials upon disposal. White and Cromartie (1985) found that disposal into an open impoundment increased the selenium content of water fowl but that there was no measurable increase in content of lead, zinc, cadmium, or mercury. Other tests have shown that ammonia, chlorinated organics, and manganese are the substances which are consistently released from suspended solids within the water column. These substances are expected to be found within the water column, near the sediments being dredged, and within the water in which the dredge spoil was disposed.

Available studies of disposal of dredged material upon land seem to indicate a minimal impact on biota, vegetation, and associated human communities. The impact of dredge disposal upon groundwater resources has not been adequately assessed in many studies. However, the impact of disposal of the dredged material upon the underlying soil can be used as an indication of the potential impact of land disposal of dredged material. In an investigation of the disposal of dredged sediments from the heavily contaminated Caluments Sag Channel near Metropolitan Chicago, Van Luik (1985) reported that transport and accumulation of metals and organics in vegetation at the demonstration site was such that the design site was an effective and safe method for land reclamation. Van Luik (1985 (b) and (c)) concluded that a substantive impact with respect to the potential long term uptake of metals and organics by vegetation was unlikely, but that careful monitoring of vegetation for metals was necessary. Van Luik also concluded that accumulation of organics would be minimal because many of the organics contained within the sediment dredge spoils would eventually degrade microbially.

In another study in which dredge materials were capped in laboratory reactors with a 50 cm thick cap composed of clay, silt and sand, it was concluded (Brannon et al., 1985) that capping prevents bioaccumulation of the dredged materials through the water column to biota. It was concluded that clay and silt represented a better barrier to accumulation of contaminants in the biota than sand did. With respect to stability of vegetation in other terestrial communities, Newling and Landin (1985) showed that plant and wildlife communities successfully developed and stabilized the dredge deposit on land.

Another method for evaluating the impacts of disposal areas is to compare contaminant concentrations in areas being dredged to Ontario guidelines for dredge spoil disposal. Most metals in sediments in areas adjoining the four main urban areas of the Bay exceed disposal guidelines (see Table 5.6.1). The metal content of other areas are unclear, pending publication of his data by R. Thomas. The problem with this approach is that guideline values are changing and the bioavailability of the metals is not characteracterized. A suggested approach is given in the next section.

Suggested Studies for Designing and Evaluating Dredging

Substantive work has been undertaken by the U.S. Army Corps of Engineers to develop management strategies for assessing environmental effects of dredging and for finalizing designs for final disposal sites (e.g., Francingues et al., 1985). Associated with such studies, have been the development of field verification programs for examining methods to predict acute and chronic toxicity of contaminated materials from dredge spoils (Gentile et al., 1985), the cytogenetic technique of sister chromatid exchange to measure genotoxic effects of in faunal polychaetes (reported by Pesch et al., 1985), changes in adenine nucleotide metabolism as indicators of stress in blue mussels and polychaetes (Zarrogin et al., 1985), and an examination of the scope for a growth index as indicators of physiological conditions of blue mussels (as reported by Nelson et al, 1985).

It is recommended that one make reference to these particular studies to develop and design a program which would minimize the impact of dredging upon both the environment being dredged and the receiving environment where the dredge spoils are to be disposed. To assess the particular impacts of dredging on the sediments and water of the Bay of Quinte, further testing and solid phase measurements of phosphorus, Kjeldahl nitrogen, organics, particularly chlorinated organics, and metal contents of the various sediment zones which may be dredged, are required. Present data are inadequate to make these assessments. It is recommended that if dredging appears to be a feasible alternative either for a localized area or for the whole Bay, then more sediment sampling be carried out to characterize the different parameters and to identify various areas of concern.

To evaluate areas which need dredging, more site specific data are also required because in order to justify dredging of only local areas, it must be demonstrated that internal

TABLE 5.6.1   U.S. EPA AND MOE BULK CHEMICAL COMPOSITION GUIDELINES FOR POLLUTIONAL CLASSIFICATION OF DREDGED SEDIMENTS

Pollution Classification	Concentration (ug/g unless otherwise indicated)				
	Non-polluted[1]	Moderately Polluted[1]	Heavily Polluted[1]	Polluted[1]	Contaminated[2]
Volatile Solids (%)	≤ 5	5-8	> 8	-	6
COD	< 40,000	40,000-80,000	> 80,000	-	50,000
Oil and Grease	< 1,000	1,000-2,000	> 2,000	-	1,500
Total Kjeldahl Nitrogen	< 1,000	1,000-2,000	> 2,000	-	2,000
Ammonia	< 75	75-200	> 200	-	100
Total Phosphorus	< 420	420-650	> 650	-	1,000
Cyanide	< 0.10	0.10-0.25	> 0.25	-	0.1
Total PCB	-	-	-	> 10	0.05
Arsenic	< 3	3-8	> 8	-	8.0
Barium	< 20	20-60	> 60	-	-
Cadmium	-	-	> 6	-	1.0
Chromium	< 25	25-75	> 75	-	25
Cobalt	-	-	-	-	50
Copper	< 25	25-50	> 50	-	25
Iron	< 17,000	17,000-25,000	> 25,000	-	10,000
Lead	< 40	40-60	> 60	-	50
Manganese	< 300	300-500	> 500	-	-
Mercury	-	-	-	> 1	0.3
Nickel	< 20	20-50	> 50	-	25
Silver	-	-	-	-	0.5
Zinc	< 90	90-200	> 200	-	100

[1] U.S. EPA (1977)

[2] MOE (Persaud and Wilkins, 1976)

loading or sediment phosphorus concentrations are excessive in these areas. Present data are lacking on this subject.

Data required include bulk chemical composition data, an interpretation of the elutriate test results, the dilution capacity of the project waters, and the transitionary nature of the dredging activities. The literature would suggest that little degradation of water quality would be expected because releases should generally be small, their effects localized and temporary, and rapid dispersion by mixing and sorption processes should limit overlying water to ambient levels. Considering the short duration of exposure, the probability of any significant bioconcentration of contaminants by fish is low. Although uptake of contaminants by benthic organisms may increase over the short-term, no net impact will likely result since these organisms are in contact over their life span with the contaminants in the sediments (interstitial waters). Accordingly, the mitigating measures which may be needed may not be excessive in conjunction with dredging.

Then toxicity and elutriate testing should be carried out to demonstrate the potential effects of dredging. The literature can be used as a guide on this subject. It indicates that there is a minimal impact in general for most metals. However, for site specific application, it is recommended that specific elutriate testing be carried out to confirm this point. The following specific types of testing would be useful:

1. Measurement of total concentrations of potential contaminants in the Upper Bay of Quinte sediments to allow comparison to US EPA and MOE guideline. (Table 5.6.1).

2. Evaluation of the sediments with the standard elutriate test. The standard elutriate test was designed to simulate the dredging and disposal process (Environmental Effects Laboratory, 1976). In the test, sediment and dredging site water are mixed in the ratio of 1:4 by volume. The mixture is shaken for 30 minutes, allowed to settle for one hour, centrifuged and filtered through a 0.45 um filter. The filtered water (elutriate water) is then analyzed for chemical constituents. Moreover, a sample of the dredging site water used in the elutriate test is filtered through a 0.45 um filter and analyzed for the same chemical constituents. A comparison of elutriate water with the filtered dredging site water for like constituents indicates whether a constituent is or is not released in the test. Dredged materials are not considered to be potentially environmentally

deleterious when the concentrations of major constituents in the standard elutriate do not increase 1.5 times above the ambient site water concentrations.

3. Conduct triplicate bioassay tests. The classification scheme would be based upon the percent mortality of the test species. Test organisms could include the mayfly nymph, Hexagenia limbata, the water flea, Daphnia magna, and the fathead minnow, Pimephales promelas.

5.7   Detrimental Effects of Sediment Burial

Sediment burial would affect two areas: the Bay of Quinte, and the area from which clean sand is dredged. The main impact on the Upper Bay of Quinte itself should be analogous to that of fly ash, except that the heavy metal content in fly ash would not be of concern. The main impacts would be the physical impact on the biota of the existing sediments, and the effect of on navigation in the shallow Bay. Burial would generally smother all existing benthic invertebrates unless they are able to migrate out of the areas being covered and then recolonize the new areas. Such rapid movements are possible for certain invertebrates but not others. Over the long-term, it is probable that a new benthic population would develop in the new sediments. However, it is unclear at what rate this would occur and whether a stable community would develop.

Another major impact of dredging would be the impact upon benthic communities in the areas dredged in Lake Ontario. However, it would be expected that the time to re-inhabitat the smaller dredged area in Lake Ontario would be much quicker than that required for re-colonization of approximately 136 km^2 in the Upper Bay of Quinte. Also, new depositional-erosional patterns devlop in Lake Ontario. They could inhibit re-establishment of benthic communities for a period of time until the lake reached a new equilibrium condition within several kilometers of the area being dredged through new sedimentation, erosion, and bottom transport processes.

A third major impact would be the effect of suspended solids and the release of any potential contaminants from the suspended solids as it settles in Lake Ontario near the dredging site or in the Upper Bay of Quinte waters. There may be an increase in turbidity by the release of light fine grain materials during the emplacement period. These potential impacts are assessed in more detail in section 5.6 in relationship to the effects of dredging and emplacement underwater of dredge spoils. The reader is asked to

make reference to the section to make in-depth assessments of various factors. In general, it is observed that suspended solids are released during dredging and during emplacement operations for a brief period of time but that these suspended solids are generally confined to near the sediments and that the elevated concentration rapidly decrease over time due to settling processes. If very fine-grain sediments are released during the dredging and/or emplacement process, then residual turbidity is prolonged for a period of time. However, even under very high velocities during dredging, elevated suspended solids rapidly, presumably due to settling and coagulation of larger particles and to dispersion processes caused by the currents.

With respect to contaminant release, the general conclusions that one could draw from assessing dredging data is that very few metals are released into the water column for a long period of time during dredging operations. The key mechanism controlling heavy metal release appears to be the fact that most trace metals and sediments are particle associated and hence, if the suspended solids (particulates) settle rapidly, then the metals are also removed. Similarly, toxic organics are not found in the water column to an appreciable extent as a result of dredging and/or emplacement operations unless colloidal material or fine-grained material remain in the water column.

These types of questions would of course be critical to any kind of dredging or emplacement operation. Elutriate testing and other chemical tests described in section 5.6 for assessing these points together with the characterization of whether there are any contaminants of potential significance in the dredged material (presumably clean fill) from Lake Ontario would need to be carried out before it could be placed in the Upper Bay of Quinte sediments.

5.8   Aeration

Increasing oxygen concentrations at the sediments will have no direct adverse effect on the benthic fauna or fish. However, the aeration device may disturb sediments.

5.8.1   Enhanced Circulation

Enhanced circulation may increase the suspended solids load (TSS). Indeed, an increase in TSS may be an unavoidable consequence of aerating sediments and may be accompanied

by an increase in total P rather than a decrease. Some of the resuspended P may not be biologically available. The increase in suspended solids may also affect biota (i.e., cover spawning beds, reduce transparency, etc.).

5.8.2 Impacts of Oxygenation

Similar to enhanced aeration, the injection of oxygen is expected to have a limited impact on the ecosystem. It could increase suspended solids due to entraining sediments if the oxygen injection system is located near to the sediment water interface. Also, as Cooke et al., (1986) note, the potential supersaturation of hypolimnetic water with nitrogen has been suggested as a problem; however, injection of only oxygen should not pose such a problem. The other major impacts are the potential increase of diffusivity processes near the sediment water interfaces. This could increase the diffusion of nutrients from sediments into the water. If this occurred, it counters the main objective of oxygenation: to minimize recirculation of phosphorus. In such a case, the diffuser would need to be redesigned to minimize disturbance of sediments.

5.9 Lake Drawdown

Drawdown could result in dewatering or drying out of exposed sediments resulting in sediment consolidation, and could result in a loss of the benthic fauna and fish. A shift in the biological community may also occur, changing the diversity and population of the benthic fauna and a loss or reduction in oligotrophic fish (salmonids).

Drawdown is infeasible. Hence, maintenance of existing water levels should have a neutral effect with respect to beneficial or detrimental effects upon the Bay of Quinte.

6.0   EVALUATION OF EFFECTIVENESS OF INTERNAL LOADING CONTROL

An experimental and modelling analysis was undertaken to evaluate the potential effectiveness of internal loading control. Two specific options were considered:

1)   the null option, and
2)   alum injection into the sediment.

This section presents the proposed model, observed fluxes, and model calibration studies. Recently obtained study results (in the past month) constrain the model calibration, but still leave major gaps in understanding. Irrespective, estimates of the effectiveness of these two control options are presented.

6.1   Proposed Model

Based upon a review of past models applied for sediment water interactions for the Bay of Quinte and other lakes, sediment fractionation measurements, and observed fluxes (see below), and the needs of a model to differentiate alum from non-alum effects, the model given in Table 1 was selected for application. Its kinetic pathways are given in Figure 1.

The model is based upon the following major considerations:

1.   SRP is transformed into algal matter (PP) in the water column and is replenished by decomposition in the water column and by sediment-water diffusion.

2.   The aerobic zone of the sediment, 2 cm thick (based upon an SOD of 2 $g/m^2/d$), is composed of: porewater orthophosphorus, labile organic phosphorus (fresh algal remains), non-labile organic phosphorus, non-apatite inorganic phosphorus (NAIP) and apatite phosphorus. It is assumed that the NAIP participates by equilibrium reactions while the apatite P is unavailable to the biogeochemical cycle for the short time period of concern here.

Figure 6.1   Definitional Sketch
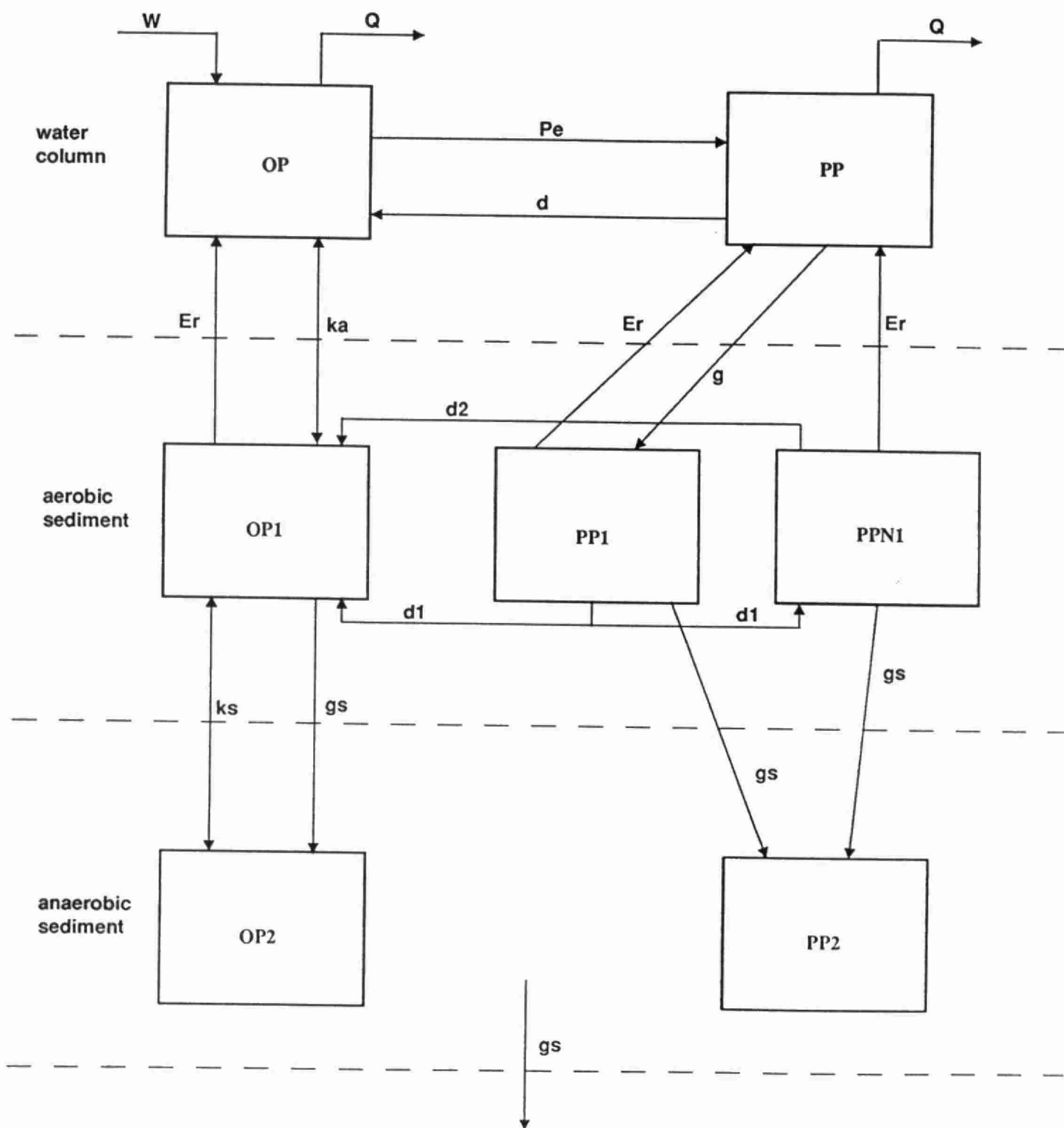
TABLE 6.1(a):   MODEL EQUATIONS

Equation
Number   Equation

(1) $\dfrac{d(OP)}{dt} = \dfrac{Lp}{Z_w} - \dfrac{(OP)}{t_w} - p_e(OP) + d_h(PP) + \dfrac{k_a}{Z_w}(OP_1 - OP) + \dfrac{E_r OP1}{Z_w}$

(2) $\dfrac{d(PP)}{dt} = p_e(OP) - d_h(PP) - \dfrac{g(PP)}{Z_w} - \dfrac{(PP)}{t_w} + \dfrac{E_r(PP1 + PPN1)}{Z_w}$

(3) $\dfrac{d(OP1)}{dt} = k_s(OP2 - OP1) - \dfrac{k_a}{ZS1}(OP1 - OP) - \dfrac{E_r(OP1)}{ZS1} - \dfrac{g_s(OP1)}{ZS1}$

$\qquad\qquad\qquad + d_1.f.(PP1) + d_2(PPN1)$

(4) $\dfrac{d(PP1)}{dt} = \dfrac{g.f(PP1)}{ZS1} - \dfrac{g_s(PP1)}{ZS1} - d_1(PP1) - E_r \dfrac{(PP1)}{ZS1}$

(5) $\dfrac{d(PPN1)}{dt} = d_1(1-f)PP1 - \dfrac{GS(PPN1)}{ZS1} - d_2(PPN1) - \dfrac{E_r(PPN)}{ZS1} + \dfrac{g(1-f)(PP)}{ZS1}$

(6) $\dfrac{dTP2}{dt} = \dfrac{gs(TP1)}{ZS2} - \dfrac{GS(TP2)}{ZS2} - \dfrac{K_s}{ZS2}(OP2 - OP1)$

(7) $fpp1 = \dfrac{E_r(PP1)/Z_w + p_e(OP)}{E_r(PP1)/Z_w + p_e(OP) + E_r(PPN1)/Z_w}$

(8) $(TP1) = (OP1) + (PP1) + (PPN1)$

(9) $(OP2) = (SVIV/(FE2)^3)^{0.5}$

TABLE 6.1b: DEFINITION OF SYMBOLS

OP   soluble phosphorus concentration in water column (ug/L)

PP   particulate phosphorus concentration in water column (ug/L)

OP1   soluble phosphorus concentration in first sediment layer (ug/L)

PP1   labile particulate phosphorus concentration in first sediment layer (ug/L)

PPN1   non-labile particulate phosphorus concentration in first sediment layer (ug/L)

TP1   total phosphorus concentration in first sediment layer (ug/L)

OP2   soluble phosphorus concentration in second sediment layer (ug/L)

PP2   particulate phosphorus concentration in second sediment layer (ug/L)

TP2   total phosphorus concentration in second sediment layer (ug/L)

W   phosphorus loading (mg/d)

GIVENS

L_p Areal phosphorus loading ($mg/m^2/d$ = 6.2 for 1978-1981)

Z_w Mean water depth (4.3 m)

k_a Sediment-water mass transfer coefficient (m/d = 0.1)

k_s Sediment-sedment mass transfer coefficient (m/d = $1.7 \times 10^{-5}/(ZS1+ZS2)$)

ZS1   Thickness of aerobic sediment layer (0.02 m)

ZS2   Thickness of first anaerobic sediment layer (i.e. second sediment layer) (0.1 m)

P_e Production rate coefficient in water column (0.2 d^{-1})

d_h Decomposition rate coefficient (d^{-1}; = 0.003 x T, where T is temperature)

d_1 Decay rate of labile particulate phosphorus (= 0.0009 T; d^{-1})

d_2 Decay rate of non-labile particulate phosphorus (= $1.13 \times 10^{-5}T$; d^{-1})

g   Settling rate of particulate phosphorus (= 0.113 m/d)

g_s Sedimentation rate

f   Labile fraction (= 0.7)

FE2   total porewater phosphorus concentration (mol/L)

SVIV   Vivanite solubility product

E_r Sediment erosion rate (8.8×10^{-5} m/d)

t_w hydraulic detention time (days)

3. The second sediment layer (i.e., the first anoxic layer) has the similar biochemical reactions occurring as in the aerobic zone. But due to the longer time scales for exchange with overlying water, it is assumed that all particulate phases (organic P, NAIP, Apatite P) can be treated as one and that their main effect is to regulate the porewater concentration of OP through the vivianite solid phase. This does not allow for formation of the subsurface SRP maxima which is observed in mid summer. The maximum is hypothesized to be caused by a kinetic effect of organic P decomposition (with increased temperature) at 10 to 15 cm coupled with the formation kinetics of pyrite (which regulates Fe(II)) and vivianite.

4. Sediment transport processes between anoxic and oxic sediments are controlled by diffusion-scale processes. The impact of oligochaetes upon transport processes is at present unclear.

6.2 Observed Fluxes

Historical estimates of phosphorus fluxes were made by Minns (1986) from a modelling study. A mass-balance model using known loadings and water column observations was calibrated based upon a settling trap measurement in a limnocorral. The main calibration parameter was the sediment-water return flux. The calibration gives estimates for sediment return of the water column of 10.7 mg $P/m^2/d$ for the upper Bay, 5.6 mg $P/m^2/d$ for the lower Bay and 9.2 mg $P/m^2/d$ for the middle Bay as an average for the period 1967 to 1981.

Recently phosphorus flux measurements have been attempted both using laboratory and field techniques. Laboratory techniques (BEAK, 1987) involved obtaining intact cores by diver, incubating the cores in the dark at constant temperature over a two-three week period and measuring P release rates both aerobically and anaerobically. The following rates were obtained.

	Aerobic Rate		Anoxic Rate	
	Mean	(Range)	Mean	(Range)
SRP Rate (mg $P/m^2/d$)	1.3	(0.1-5.3)	2.8	(0.02-7)
PP Rate (mg $P/m^2/d$)	1.3	(0-3.2)	1.2	(.05-3.7)
TP Rate (mg $P/m^2/d$)	2.3	-	3.93	-

The measurements indicated that there was an increase in both the SRP and the TP (total phosphorus) concentration over time. But experimental conditions ensured that there was no physical erosion of particulate sediments. Degassing by bubble ebuliation could erode particulate matter, but no bubble formation was evident. It was suggested that SRP diffused from the sediments and accumulated in particulate matter in the water column with the particulate P averaging about 40% of the total P flux to the sediments. These measurements averaged approximately 20-25% of the value obtained by Minns and were quite variable and episodic.

The laboratory cores showed no apparent biogenic activity. This suggests that biogenic reworking was not significant. In addition, a BEAK (1987) test procedure would underestimate true phosphorus efflux. Phosphorus efflux is driven by continuous algal settling from the water, degradation, and consumption reworking at the sediment-water interface. Removal of sediment cores from the environment result in decreased fluxes to the sediments and an associated slowing down of efflux over time. The modelling assessment presented below could be used to estimate the change in efflux which occurred in the laboratory measurements.

Interstitial phosphorus concentrations were measured in the Bay of Quinte by Manning (personal communication) and BEAK (1987). Based upon interstitial water (peeper profile) data from one site, the diffusional gradient from the top 2 cm suggested a diffusional flux from the sediments of 11 mg $P/m^2/d$ for a period in late summer. A re-analysis of the data using information from the top 6 cm over which the main gradient occurs would suggest a diffusional flux of 2 mg $P/m^2/d$. The higher value is more appropriate because the gradient is sharper in the top and decreases over the depth range of 2 to 6 cm.

Sly (1988), in an imaginative but potentially controversial approach, measured internal loading by comparing the following 3 fluxes: (i) phosphorus accumulation in a dialysis chamber from settling seston which decomposed; (ii) phosphorus accumulation in a dialysis chamber from sediment decomposition and diffusion; and (iii) SRP accumulation in dialysis chambers facing horizontal currents. The measurements are difficult to extrapolate with confidence because the interpretation is dependent upon (i) the settled seston decomposing at rates similar to those at the sediment-water interface, and (ii) representative capture of the decomposed material by the dialysis membrane.

Ignoring these difficulties, his data do provide a useful quantitative basis for interpreting the probable magnitude of internal loading. His data are especially useful for differentiating the relative sources of the internal load between porewater diffusion and decomposition of fresh seston. The following results were obtained.

	Phosphorus Release $(mg\ P/m^2/d)$		Oxygen Consumption $(g\ O_2/m^2/d)$	
	Seston Decomposition	Sediment Diffusion	Seston Decomposition	Sediment Diffusion
Picton Harbour (21 wks. of measurement)	28	0.93	2.2	2.1

	Phosphorus Release $(mg\ P/m^2/d)$		Oxygen Consumption $(g\ O_2/m^2/d)$	
	Seston Decomposition	Sediment Diffusion	Seston Decomposition	Sediment Diffusion
Picton Harbour (14 wks. measurement)	28	-3.7	1.4	0.6
Hay Bay (14 wks. measurement)	42	9.3	1.2	0.65

The negative sediment diffusion for the Picton Harbour indicates net diffusion into the sediments. The 21 week period includes spring/summer/fall while the 14 week period includes later summer/fall. The 21 week data was interpreted to mean that internal loading was generated 97% from fresh seston and 3% from bottom sediment diffusion. The 14 week data for Picton Harbour were interpreted to mean that 100% of the internal loading was derived from seston with 14% of the settling flux accumulating in the sediment through water to sediment diffusion while the 14 week data for Hay Bay were interpreted to indicate that 80% of the internal loading was derived from settling seston and 20% from a sediment diffusive efflux. In another set of measurements at Hay Bay and Picton Harbour designed to check measurement variability, the settling seston generated 78% and sediment diffusion 22% of the internal loading. One measurement by Sly (Sediment efflux for Hay Bay, 14 weeks) of 9.3 mg $P/m^2/d$ agrees quite well with the calibrated backflux of Minns (1986).

The estimates of internal loading by Sly (1988) vary from approximately 28 mg $P/m^2/d$ at Picton Harbour to 50 mg $P/m^2/d$ at Hay Bay. These values are within a factor of 5 of those of Minns calibration but approximately 10 times those of Andrews (1988). It is possible that Andrews' estimates overall are low because obtaining cores just after an accumulation of a pulse of fresh seston is difficult. His wide range suggests the possibility of such pulses. By being present continuously, Sly's technique integrates these pulses. However, it is difficult to see Sly's estimates as being that much higher.

Overall, the estimates of Sly are substantially higher than those of the others. But the measurements are actual and must be taken as real. Due to the large benthic invertebrate population upon the outer membranes of the dialysis chamber, ingestion of fresh seston must be occurring. Thus, his technique is capturing a high turnover flux which occurs within 10 cm or so of the sediment-water interface. Such fluxes are not captured by other techniques.

Inspection of Sly's "SOD" data suggests plausible values (typical values for such sediments are of the order of 1 $g/m^2/d$), but they are substantially higher than historical measurements (0.3 $g/m^2/d$). The technique of injecting the tube into the sediments would result in some sediment disturbance which could make the sediment diffusion measurement somewhat high. But, this would not affect the "Seston Decomposition" SOD. Also, because the sediments are quite soft, because redox profiles re-establish themselves relatively quickly, and because tube injection was done quite carefully, it is doubtful that the SOD is excessively influenced by the tube injection.

Another piece of data which aids interpretation of Sly's data are sediment pore water profiles of SRP and ammonia measured by Manning (personal communication). The data show a subsurface maximum of 15 cm of depth for SRP and ammonia for Big Bay but a uniform profile for the top 10 cm of the Hay Bay site before an increase in SRP is observed. The Big Bay site indicates a pore water diffusion flux of 2.1 mg $P/m^2/d$ for the top 3 cm and 1.9 mg $P/m^2/d$ over the top 10 cm. The Hay Bay site indicates no pore water diffusion over the top 10 cm. If it is assumed that sediment mixing is keeping the SRP and ammonia profiles in Hay Bay uniform over the top 10 cm, the SRP gradient between 10 and 20 cm would imply a deep pore water diffusional supply of 0.6 mg $P/m^2/d$. This value is substantially below that from Sly's estimate for Hay Bay.

Another possible explanation for the lack of diffusional profile is methane gas ebulation, recently modelled by DiToro et al. (1988). Ebulation occurs naturally in Hamilton Harbour (Snodgrass, 1977). The possibilities of ebulation requires future assessments.

6.3   Synthesis of Flux Data

The synthesis of the above data is fraught with many pitfalls and other geochemical influences. Based upon Minns (1986) modelling analyses, the following fluxes occur:

1) Loading from land based sources   = 6.2 mg $P/m^2/d$.
2) Outflow (45.6 ug P/L, q_s = 0.154 m/d)   = 7.0 mg $P/m^2/d$
3) Internal load   = 10.7 mg $P/m^2/d$
4) Settling flux   = 9.9 mg $P/m^2/d$

The settling flux infers a settling velocity for total phosphorus of 0.22 m/d (79 m/yr). It represents a water column average, rather than a value for the bottom 0.5 m above the sediment.

The flux estimates of Sly due to diffusion from shallow (1-2 cm) sediments are sufficiently variable (-4 to 10 mg $P/m^2/d$) to suggest that his range may be an indication of uncertainty associated with his methods. This data range may also reflect patchiness of the true sediment surficial layer due to particle erosion, especially in Hay Bay. Due to the variable peeper pore water data of (Andrews and Manning), there are not sufficient data to un-ambiguously interpret pore water diffusion characteristics. Because the one core of Andrews and the Big Bay core of Manning show similar characteristics, their data are taken in this study to indicate pore water diffusion. Assuming a several centimeters gradient, the pore water diffusional flux of SRP is of the order of 2 mg $P/m^2/d$.

The calibration of Minns would suggest a reasonable value for recycle from sediments. However, it is based upon an estimate of particulate P settling. The values of Sly for seston decomposition are reasonably constant for Picton Harbour. But more erosion is suspected to occur in Hay Bay than in Picton Harbour (Sly, personal communication). Assuming that sediment erosion occurs at all sites of Sly and that more eroded material is carried vertically a few centimeters above the sediment than several meters above the

sediment, the flux estimated by Sly is plausible. Accordingly, the conceptual picture involves sediment erosion and "seston" (as labelled by Sly) decomposition within an 0.1 to 0.5 m zone of the sediment-water interface equivalent to 30 mg $P/m^2/d$ (using Sly's Picton Harbour data as the base case). Such a value would be an underestimate of total sedimentation because this value incorporates decomposition, a process which would involve only a portion of the settling flux from the water column.

For the purposes of this study, the following interpretation of the above data is adopted.

1. The net internal loading from the sediments is 30 mg $P/m^2/d$ (based upon Sly's data). This is the softest number of the whole group described below. It is probably ±100%.

2. The net sediment accumulation is 1.4 mg $P/m^2/d$. Accordingly the net difference between land-based inflow and water column outflow to the Middle Bay must be this value, or the net difference between sediment recycle and settling must be this value. Because land based loadings and water column concentrations are known relatively precisely, the net differences must occur in the vertical fluxes.

3. The outflow loss is 7.0 mg $P/m^2/d$ (based upon C = 45.6 ug P/L; q_s = 0.154 m/d). The inflow loading is 6.2 mg $P/m^2/d$ (Minns, 1986).

4. These values define the settling flow (i.e. the sediment loading as 29.25 mg $P/m^2/d$). This results in the sediment fluxes being out of equilibrium with a net loss of 2.2 mg $P/m^2/d$.

5. Of the 30 mg $P/m^2/d$ internal loading, 2 mg $P/m^2/d$ is taken to be pore water diffusion. Hence, pore water diffusion is 7% of the total internal loading, a value consistent with some of Sly's data. (An alternative interpretation of internal loading as 10 mg $P/m^2/d$ based upon Minns (1986) analysis would place sediment-water diffusion at 20%.) As noted above, this estimate for diffusion may be low if significant oligochaete activity or methane gas ebulation occurs locally.

6. Of the sediment loading, of 29.2 mg $P/m^2/d$, 12.1 mg $P/m^2/d$ is taken to be due to settling of seston, consistent with the value of Minns (10.7) but including the effects

of net sedimentation (1.4). Hence, 17.1 mg P/m^2/d represents erosion of sediment phosphorus by bottom currents and a short cycling distance of approximately 0.5 m above the sediment-water interface.

7. The sediment-water flux of P measured by Andrews (2.3 mg P/m^2/d) is sufficiently close to the pore water diffusion value to suggest (i) that it integrates some fresh seston accumulation in the sediment, but (ii) that it mainly represents sediment decomposition of seston out of the surficial 2 cm of sediment.

8. The sediment dry weight flux and organic carbon settling flux were respectively:

   i)   at Picton Harbour - 36 g dry weight/m^2/d and 15 g C/m^2/d; and
   ii)   at Hay Bay: 200 g dry weight/m^2/d and 16 g C/m^2/d.

Based upon the dialysis fluxes of 27.9 mg P/m^2/d and 42.8 mg P/m^2/d respectively, the following ratios results:

   i)   at Picton Harbour,   P/OM = 0.8 mg P/g OM
                               P/C = 1.9 mg P/g Carbon

   ii)   At Hay Bay,   P/OM = 0.2 mg P/g OM
                               P/C = 2.7 mg P/g carbon

Based upon the potential of 1.2 mg P/g sediment, the Picton Harbour dialysis tube infers that approximately 70% of settling P decomposed at Picton Harbour and 18% of settling P decomposed at the Hay Bay site. These ratios are extremely high, suggesting that the dialysis data may be suspect.

Another explanation is as follows. These values imply that only a small portion of the Picton sample is due to erosion and deposition but that much more of the Hay Bay sample included resuspended and partially decomposed material. Decomposition of the Hay Bay sample is sufficient to release phosphorus but not to alter the C:N ratio in the remaining particulate matter. This is supported by the quite consistent C:N ratios and the lack of sediment colour change which would occur after pulse loadings. This interpretation also suggests that Picton Harbour acts like a giant "incubator" in which decomposition rates must be high or else the sediment P content would be very much greater than in Hay Bay.

6.4   Model Calibration

The model parameters were calibrated using the following logic:

I   For initial conditions, the total phosphorus concentrations is 45.6 ug P/L of which 35 is assumed to be soluble and the remainder particulate. Of the 290,000 mg P/m^3 in the sediment (based upon 1.2 mg P/g (Manning, 1988), a porosity of 0.9, and a solids density of 2.45 (Minns et al., 1986), 50% is organic, 25% is NAIP and 25% is Apatite P (Manning, 1988).

II   The erosion rate (E_r) is estimated as follows. Decomposition of the erosion flux over a week period results in a flux of 17.1 mg $P/m^2/d$. This gets recycled and readily decomposes outside the dialysis membrane. The most readily available source of the recycle is NAIP. Organic P will decompose, but at a much slower rate than equilibrium minerals. Accordingly, it is assumed that the decomposition of the erosive flux originates mainly from NAIP.

In the water column, the erosive flux will decompose. If the material returns to the sediments, it is doubtful that it will decompose but rather that it will rapidly regain equilibrium with the sediment pore water.

Based upon a sediment NAIP plus organic matter content of 0.9 mg P/g (220,000 mg P/m^3 of sediment), the erosive rate coefficient (E_r) is 8.8×10^{-5} m/d. Based upon the settling flux in Sly's traps of 25 mg $P/m^2/d$, and the assumptions that this is all derived from erosion, E_r is 1.8×10^{-4} m/d.

A decomposition rate in the water column for this material of 0.01 d^{-1} is assumed as an integration of rapid equilibrium mineral dissolution and slow organic matter decomposition. A settling rate for this material of 0.6 m/d is assumed.

III   The particulate phosphorus production rate is 0.2 d^{-1}; the particulate phosphorus decomposition rate in the water column is 0.003 T d^{-1}.

IV   The settling rate of particulate phosphorus produced in the water column is 0.113 m/d based upon the modelling estimate of Minns (1986).

V   The sediment efflux mass transfer coefficient has a value of 0.2 m/d, based upon analysis of the limnocorral tracer data of Hesslein (Snodgrass, 1986)

VI   Based upon the modelling analysis of algal decomposition dynamics in the sediments of Lake Erie, the following coefficients were adopted. Algae decomposition occurs at the rate of .0108 d^{-1} at 12°C. 0.7 of the algal decomposition is liable, resulting in direct back generation to the water column. The remainder of algal P enters the organic P pool which decomposes at the rate of 1.35×10^{-4} d^{-1} at 12°C.

6.5   Testing of Model Calibration

With this calibration, the model calculations were first tested assuming:

a)   that sediment algal decomposition cycles directly into the interstitial pore water of the top box; and

b)   that sediment algal decomposition cycles directly into the overlying water column.

Based upon that analysis, the concepts given in the equation of Table 6.1 were retained (i.e. algal decomposition cycles directly to the interstitial pore water and then diffuse out of the sediment.

The results of the calibration are given in Figures 6.1a to 6.1f. The simulations indicate that soluble phosphorus is converted to particulate phosphorus over the season and that less is formed in the cooler months. The interstitial pore water concentration builds up to approximately .13 mg P/L. This is close to, but somewhat lower than observed values in the vicinity of 0.2 to 0.3 mg P/L in the top 2 centimeters of the sediments. To achieve this, the labile phosphorous decomposition coefficient was incremented by 6 times that of DiToro and Connolly (1980). Some further studies are required to evaluate this aspect of the model.

The labile and refractory forms of organic carbon (Figures 6.1d, e) and total phosphorus in the anaerobic sediment (Figure 6.1f) follow typical cycles which are expected. However, there are no data available to test this calibration.

Fig. 6.1a Soluble Phosphorus in Water Column

Fig. 6.1a Soluble Phosphorus in Water Column
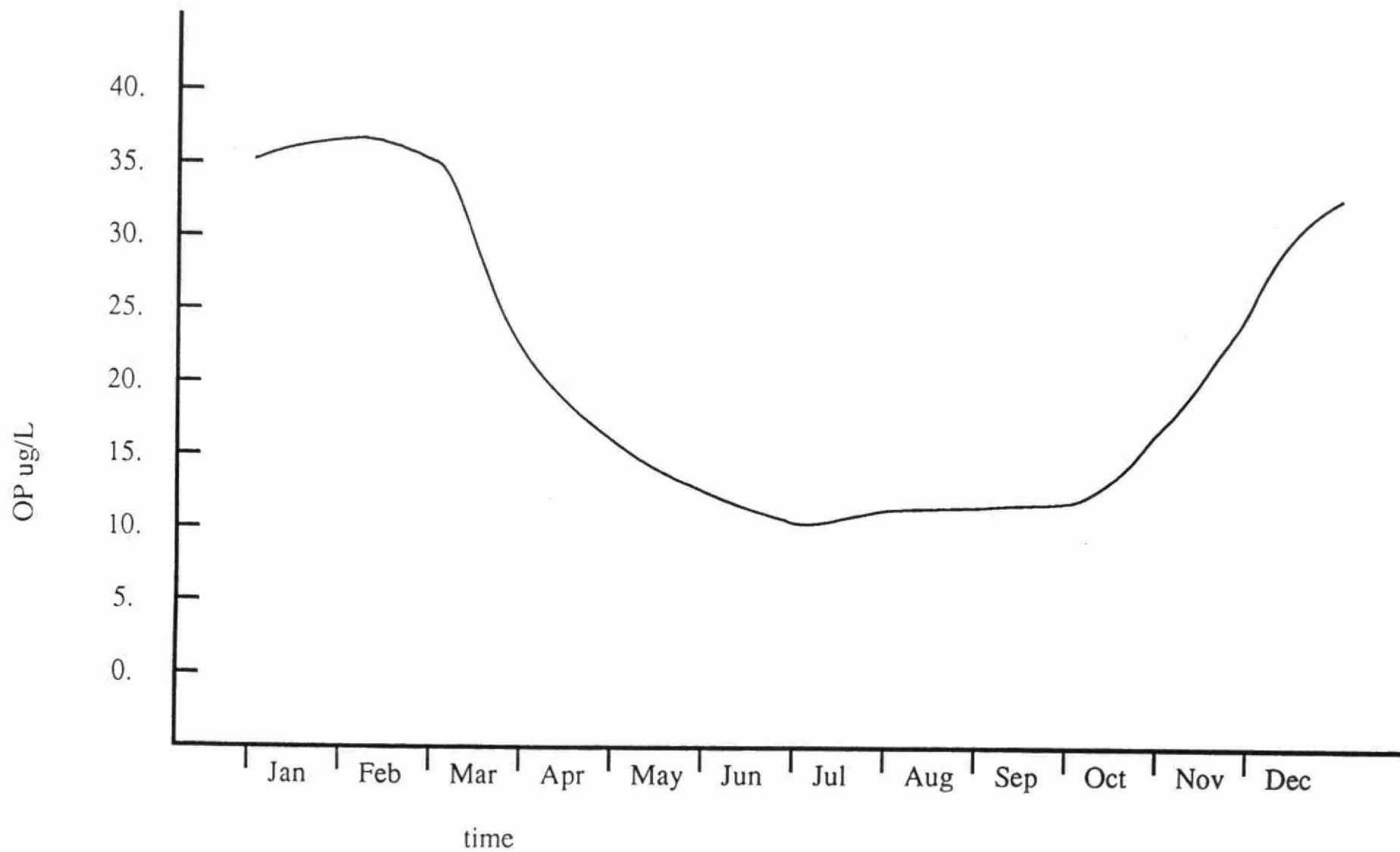
Fig. 6.1b Particulate Phosphorus in Water Column
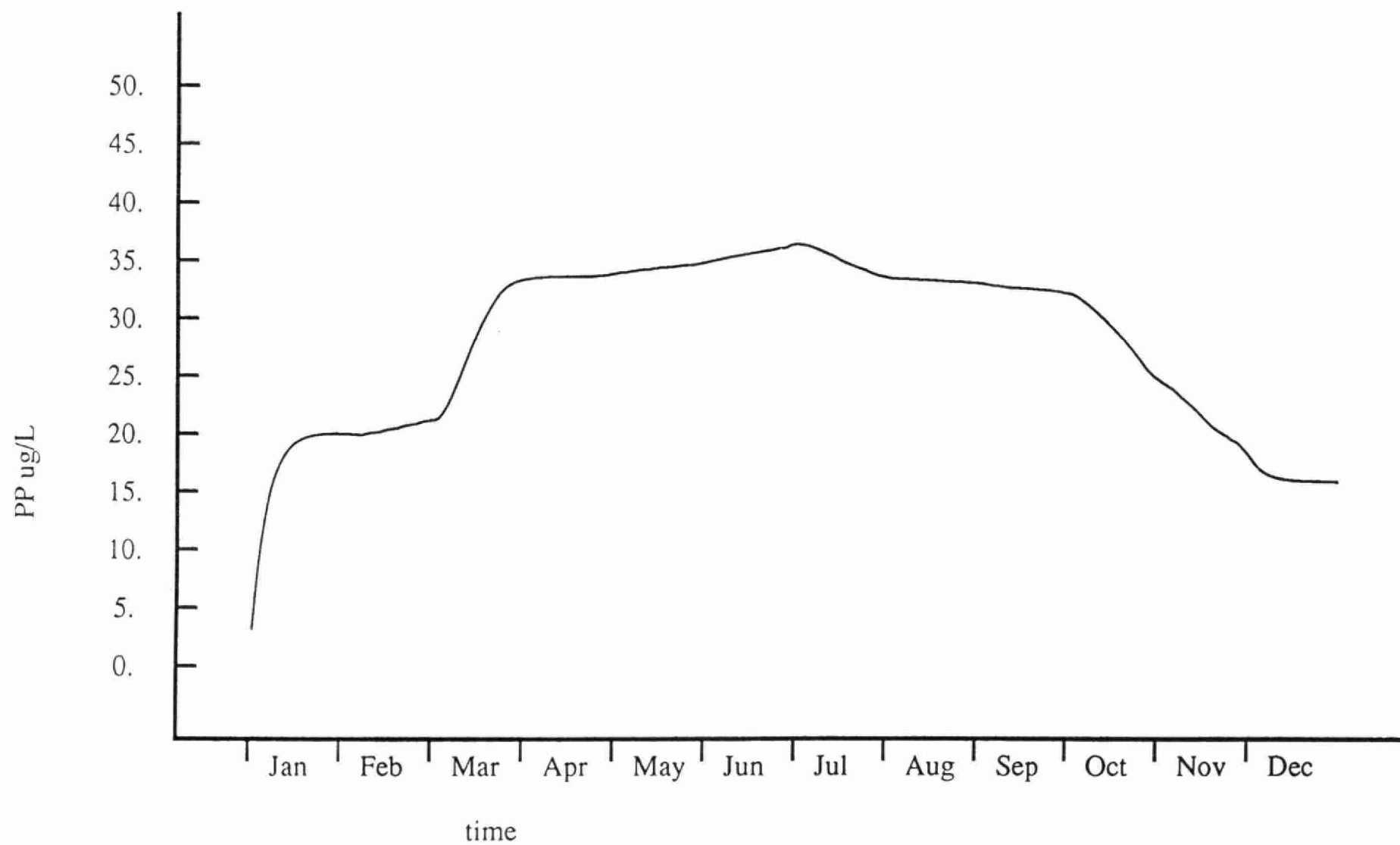
Fig. 6.1c Soluble Phosphorus in Aerobic Sediment
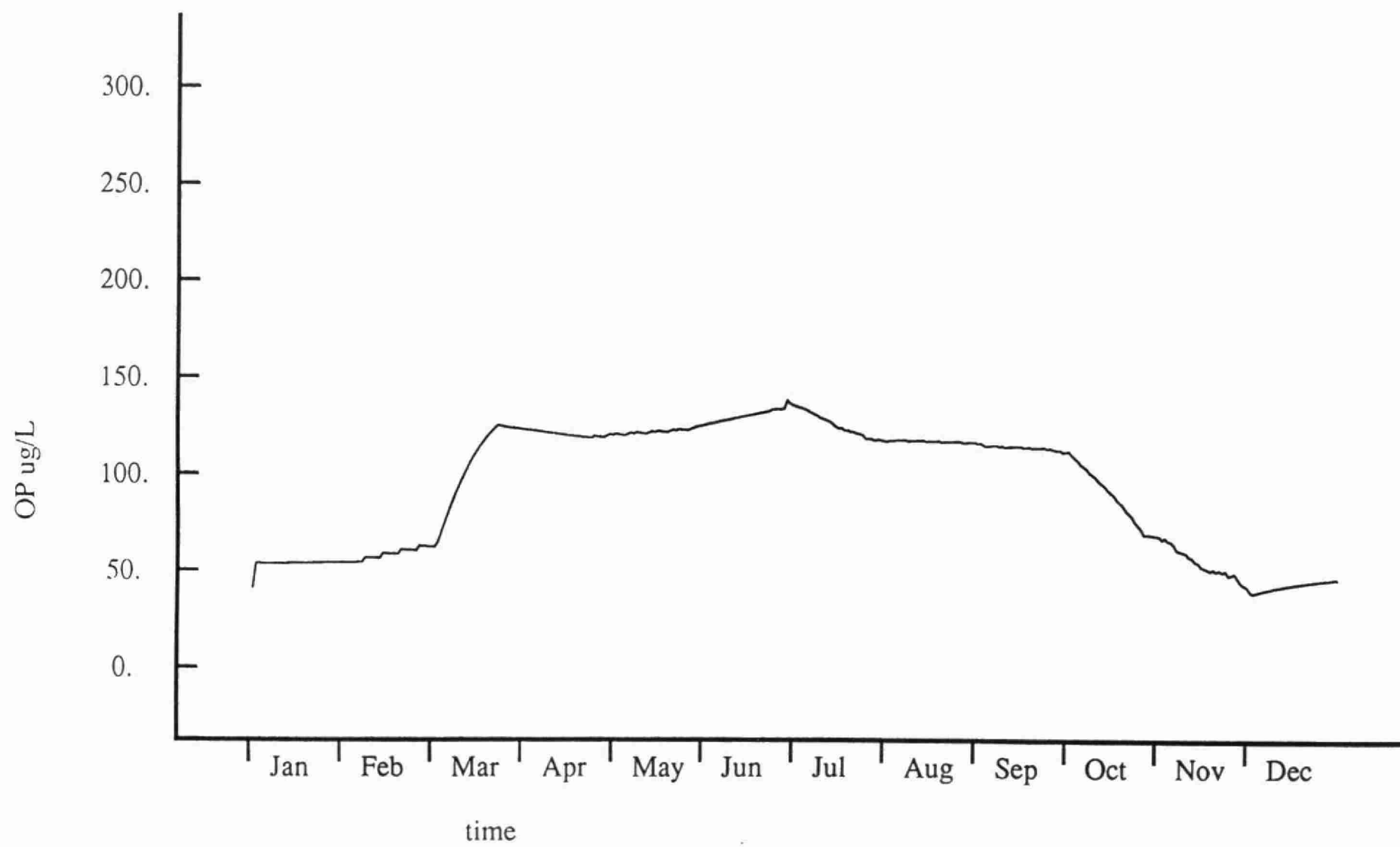
Fig. 6.1d Labile Particulate Phosphorus in Aerobic Sediment
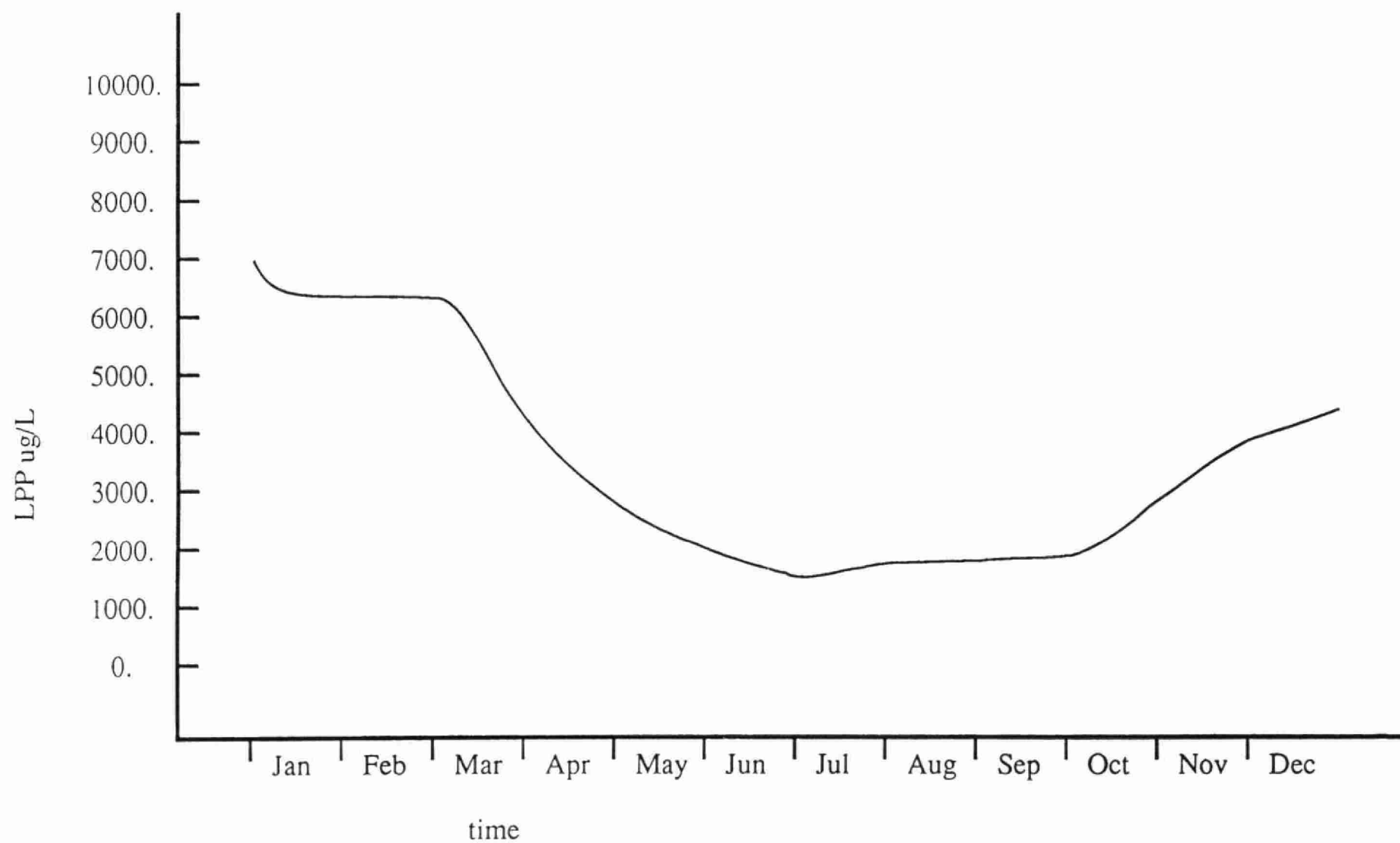
Fig. 6.1e Non-labile Particulate Phosphorus in Aerobic Sediment
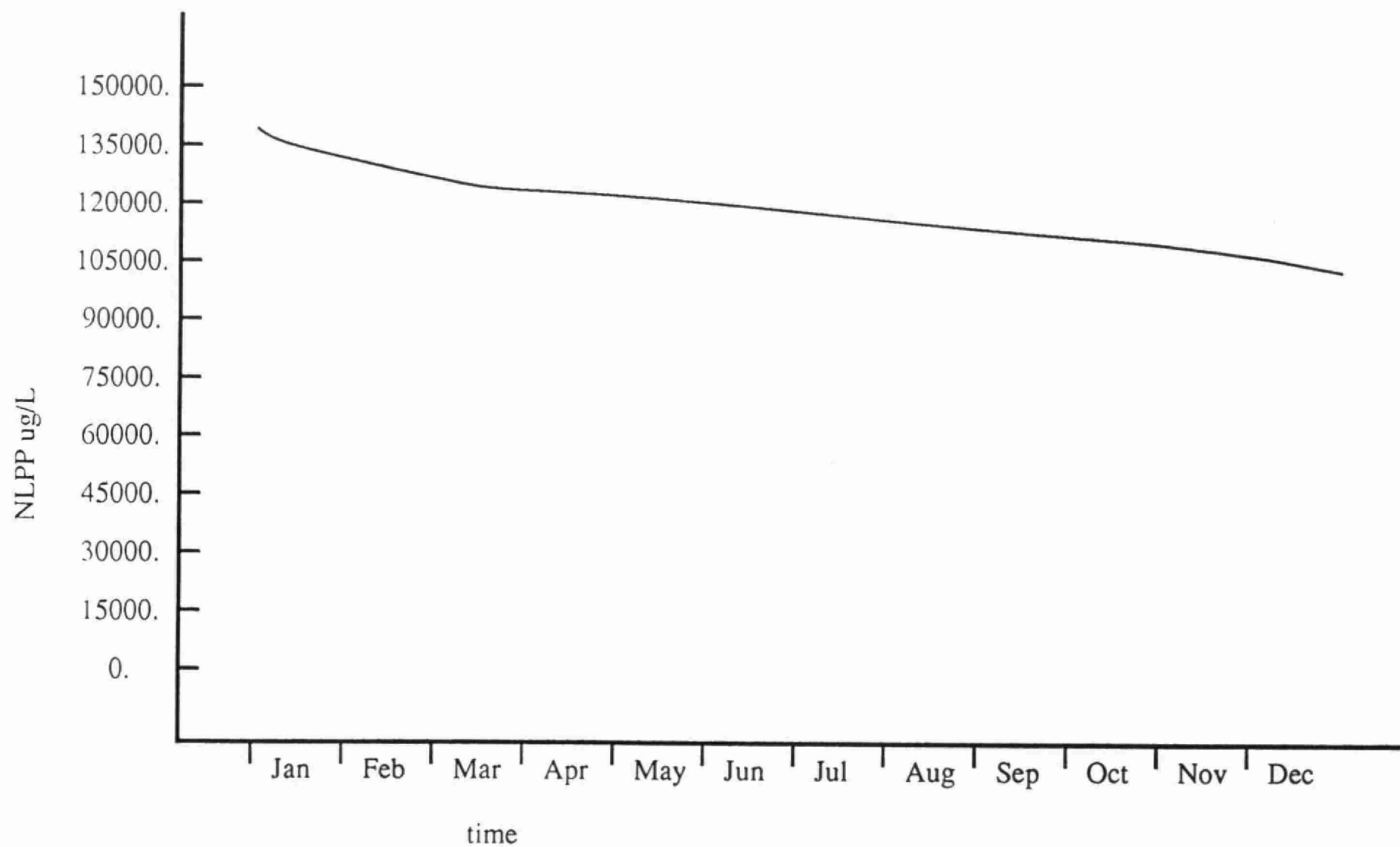
Fig. 6.1f Total Phosphorus in Anaerobic Sediment
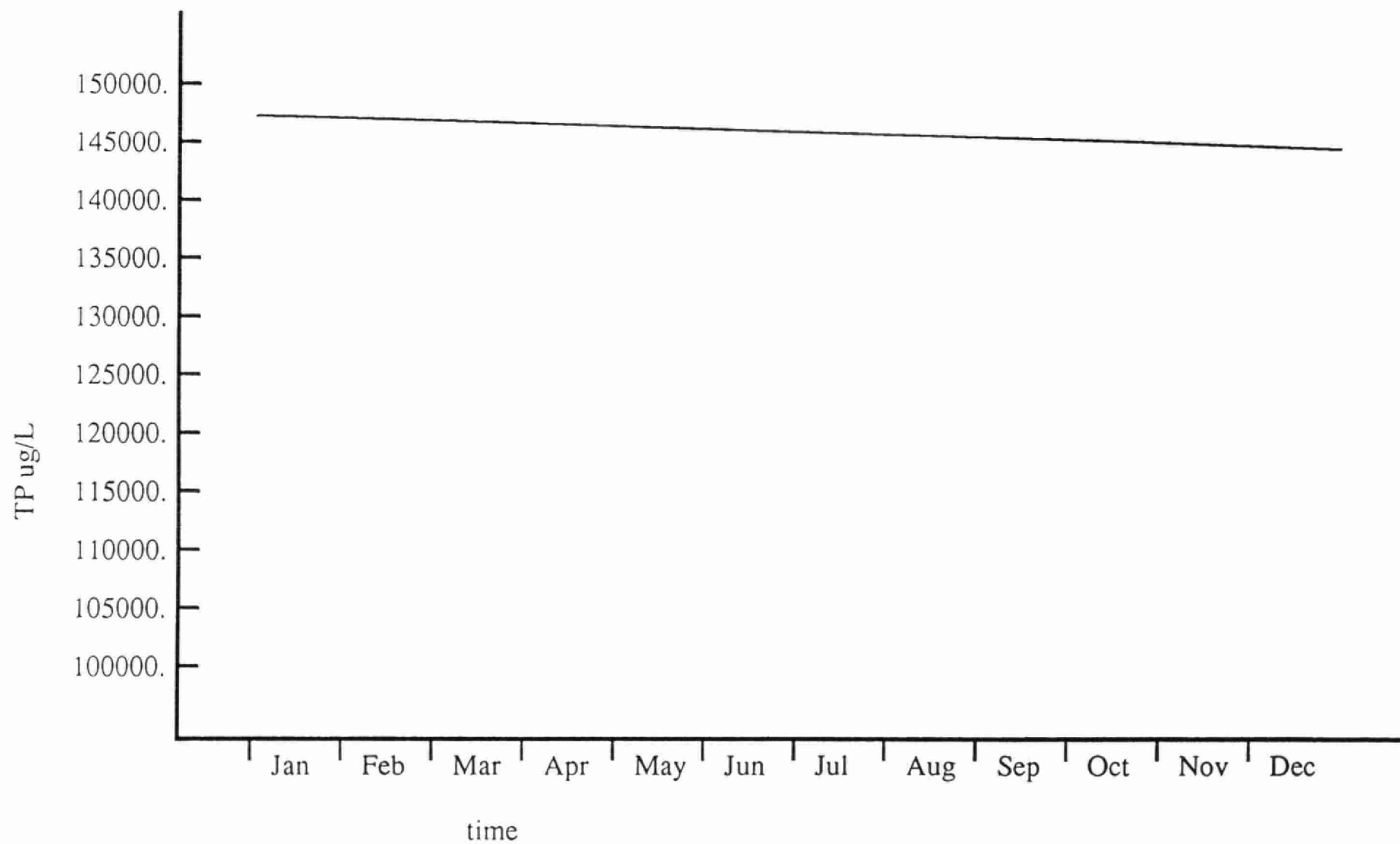
Fig. 6.1g Total Phosphorus in Water Column
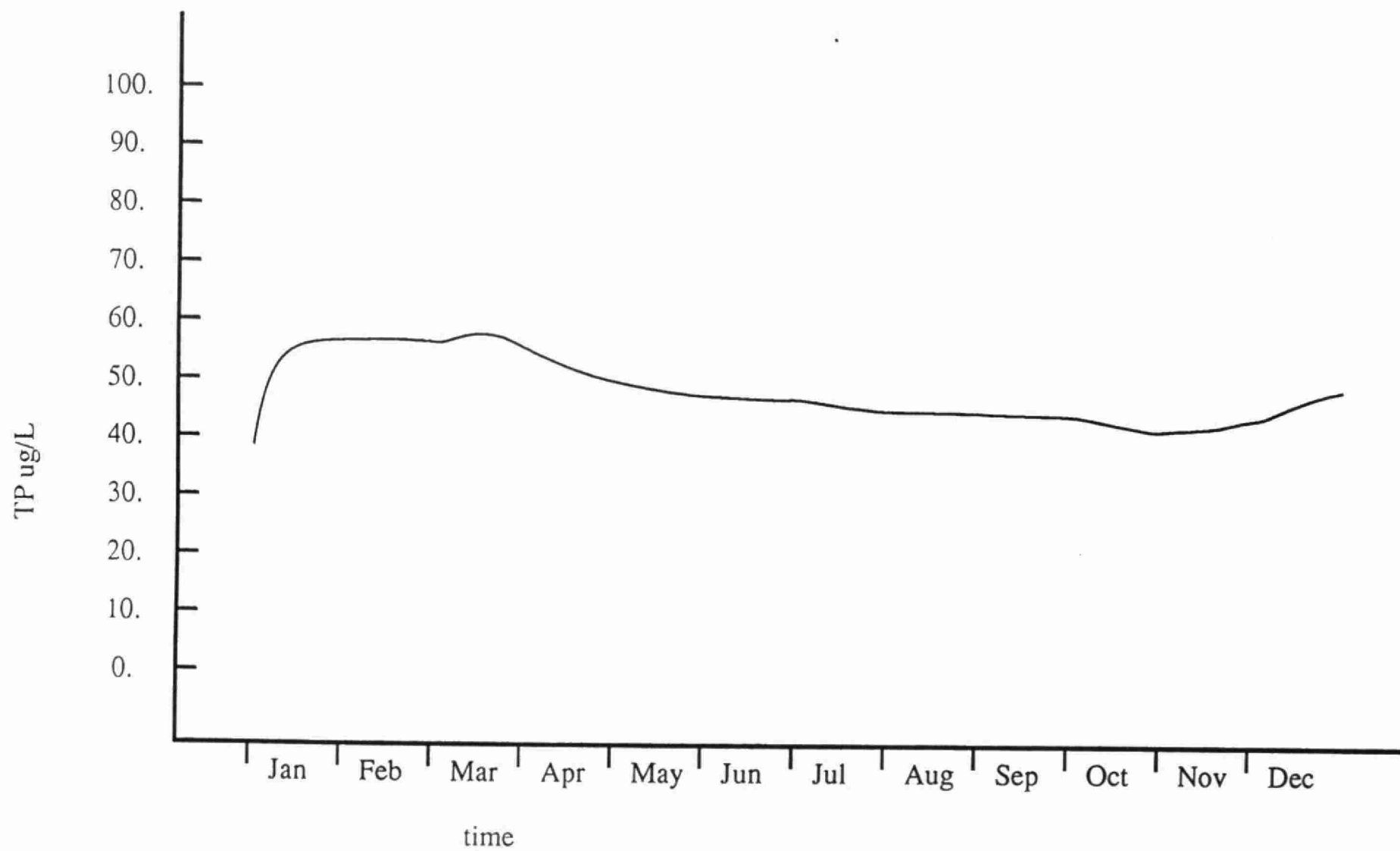
The major final check on the calibration are the fluxes (mg P/m²/d) calculated by the model. They are as follows:

	Model Calculations	Field Observations
Anaerobic sedimentation rate (mg P/m²/d)	1.6	1.6
Aerobic sedimentation rate (mg P/m²/d)	1.3	-
Diffusive flux to water (mg P/m²/d)	11	2 and up*
Internal loading (mg P/m²/d)	22	10*-30*
Aerobic zone porewater concentration (mg/L)	0.13	0.2-0.3
Settling flux (mg P/m²/d)	22	22
Net (settling flux - back flux) (mg P/m²/d)	-0.63	-0.8
Total phosphorus in water (ug/L)	45.	45.6
Soluble phosphorus in anaerobic pore water	1.9	1.8-2.4

*Range of values dependent upon investigation.

The values given above are summer averages. Overall, the values agree reasonably with observation suggesting that a reasonable, self-consistent calibration has been attained.

Further work should be carried out to include more sediment boxes to better describe the deep sediment profile.

A sensitivity analysis of the influence of land based loading and residence time upon internal loading was also undertaken. Significant variations in the flushing pattern occur seasonably, particularly due to lake-bay exchange and annual hydrological cycles. The analysis indicates that internal loading model parameters have a larger effect upon water column concentration than seasonal variations in loadings. However, over the long term, alterations in land-based loadings or permanent changes in the flushing start to influence the water column concentrations.

6.6 Aluminum Precipitation/Coagulation Experiment

6.6.1 Methods

Sediment samples from the cores obtained by Andrews (1988) and used for measuring sediment efflux from the Upper Bay of Quinte were sliced from the surficial 1 cm of several core tubes sufficient to provide enough mass for experimentation. The sediment slices were homogenized and mixed. 10 mL of the sediment sample (porosity .965) were added to each of 8 flasks containing 100 mL of Upper Bay of Quinte water. The resultant loading was equivalent to 8.5 g sediment/L and 10 mg P/L.

Four flasks were used as controls and four for experimental addition of aluminum (5 mg Al^{3+}/L plus 0.6 meq. $NaHCO_3$/L to neutralize the aluminum acidity). Two control and two experimental flasks were shaken for 1 hour while the other two controls and two experimental flasks were shaken for 24 hours. After completion of shaking, an aliquot was filtered and total dissolved phosphorus and aluminum measured. After settling for one hour, total dissolved aluminum and phosphorus and total phosphorus and total aluminum were measured. Phosphorus was measured on all samples while aluminum and pH were measured on representative samples.

6.6.2 Results

The raw experimental results are given in Table 6.2a. The duplicate values are summarized in Table 6.2b. The results indicate the following:

i) There was a small decrease in pH from the ambient sediment pH of 7.3 to the mixture pH of 6.7 to 6.9. The pH was the same in the controls as in the treatments, resulting in a minimal impact upon the formation of aluminum hydroxide flocs.

ii) The residual dissolved aluminum concentration after aluminum addition is essentially the same as that of the controls which had no aluminum addition. This suggests that the ambient pore water had no aluminum addition. This suggests that the ambient pore water is at equilibrium with respect to aluminum hydroxide.

iii) There is an increase in TDP released to the water from the particulate phase in the controls between 1 hour and twenty-four hours of shaking. The data suggest no increase in TDP in the aluminum injected solids immediately after shaking but a similar increase in TDP after settling.

iv) There is a slightly smaller amount of total phosphorus left in the water after settling due to alum injection than without alum treatment. This suggests that alum aids in coagulating some of the particles.

v) The solid phase after settling has somewhat less aluminum in the treated flasks than in the controls, despite the addition of 5 mg/L Al in the treated flasks. The differences, however, are not large.

Thus, overall these experimental results suggest that there may be some release of dissolved phosphorus due to shaking and decomposition over a 24 hour period but that the release is not large. Addition of alum will precipitate soluble phosphorus, but the addition in this experiment was not large enough to induce a further reduction in soluble phosphorus levels (except possibly immediately after shaking). At a ratio of 5 mg Al^{3+} to 9 gm of sediment, the majority of the aluminum presumably coagulated some of particulate phase and had insufficient capacity left over to impact SRP levels. Further experimental work is required to evaluate proper dosages and the effect of alum upon phosphorus control.

6.6.3 Implications for Modelling Assessment

Equilibrium calculations of aluminum and phosphorus were undertaken using an equilibrium computer code. Phosphorus control may occur by precipitation of aluminum phosphate (variscite) or by precipitation of aluminum minerals such as amorphorus aluminum hydroxide or gibbsite onto which phosphorus adsorbs.

Calculations assuming the simultaneous formation of aluminum hydroxide and varisite ($AlPO_4 \cdot 2H_2O$) indicate that $Al_T = 10^{-4.12}$ and $TP = 10^{-5.84}$ (.05 mg/L) due to formation of the amorphorus $Al(OH)_3$ phase and that $Al_T = 10^{-6}$ and $TP = 10^{-4.09}$ (3 mg/L) for the gibbsite phase at the sediment pH (7.25). For anoxic pore water, formation of vivianite results in equilibrium concentrations being $Fe_T = 2$ mg/L (3.5×10^{-5} mol/L) and $TP = 0.08$ mg/L (3×10^{-6} mol/L).

Observed iron concentrations in the anaerobic pore water are of the order of 2 mg/L while observed TP concentrations are 0.25 mg/L in the aerobic zone and 2 mg/L in the anaerobic zone. This suggests that the pore water is supersaturated with respect to virasite during the summer months. This would suggest that the equilibrium model given in Table 6.1 will need some modification to account for these kinetic concepts.

There is an additional interpretation of this data which should be attempted, but was not in this present study - the role of coprecipitation in the laboratory and in nature. The high primary productivity in the bay results in biological induced chemical precipitation of calcite. Coprecipitation of phosphorus usually occurs simultaneously. Sediment data imply that calcium:phosphorus ratios have recently increased. Further equilibrium calculations, mineralogical studies and in situ measurements of pore water alkalinity and pH are required to assess this possibility.

The effect of aluminum upon phosphate levels will vary over more than an order of magnitude depending upon the solid phase present. Assuming that amorphorus aluminum hydroxide first forms, and that it retains its effectiveness for three years, the interstitial pore water concentration of phosphorus in the aerobic zone would be limited to of the order of 0.1 mg/L by aluminum coagulation/precipitation. This assumption is employed in the modelling of the null option.

6.7   Effectiveness of Control Options

6.7.1   Null Option

The model calculations for the null option are given in Figures 6.2 for Year 3, in Figure 6.3 for Year 5, and in Figure 6.4 for Year 10 assuming that the present land-based loading continues at a constant rate but that the sediments attempt to approach a steady-state with the overlying water. The calculations indicate that the sediment has approached close to a steady-state within several years of the initial conditions. Similar calculations could be made for changes in loading to evaluate the absolute response of the sediment expected. A full model validation exercise should, however, be undertaken to evaluate Bay of Quinte data from the mid 1970's to present before these calculations should be used with much confidence.

		Concentration After Settling		Before	Concentration Dissolved		Total	
Flask*	Shaking Period	TDP	Shaking Al	pH	TDP	Al	TP	Al
C1	1 hr.	.017	.08	6.81	.015	.08	0.019	3.5
C2	1 hr.	.023	–	6.87	.013	–	0.22	–
C3	24 hr.	.039	–	6.85	.074	–	0.24	–
C4	24 hr.	.078	.14	6.90	.103	.10	0.23	4.6
Ex1	1 hr.	.023	.12	6.71	0.072	0.24	.136	2.7
Ex2	1 hr.	.007	.12	6.85	0.012	0.10	.135	2.9
Ex3	24 hr.	.027	.12	6.79	0.077	0.08	.131	2.8
Ex4	24 hr.	.013	.12	6.77	0.084	0.08	.152	3.4

TABLE 6.2a: EXPERIMENTAL DATA FROM ALUMINUM ADDITION

*C = Control
Ex = experimental
Units: P and Al = mg/L
TDP = total dissolved phoshporus
TP = total phosphorus

TABLE 6.2b: SYNTHESIZED EXPERIMENTAL DATA

Treatment	Concentration After Shaking		Concentration after Settling			
	TDP (ug/L)	Diss. Al (mg/L)	TDP (ug/L)	Diss. Al (mg/L)	TP (ug/L)	Total Al (mg/L)
Control, 1 hr. shaking	20	.08	14	.08	200	3.5
Control, 24 hr. shaking	60	.14	89	.10	230	4.6
Experimental, 1 hr. shaking	20	.12	42	.17	136	2.8
Experimental, 24 hr. shaking	20	.12	80	.08	141	3.1

TABLE 6.3:   CALCULATED FLUXES FOR DIFFERENT CONTROL OPTIONS
(Units: mg/m^2/d)

| Flux | NULL OPTION | | | | Alum Control* | Erosion Control* |
	Year 1	Year 3	Year 5	Year 10		
Anaerobic sedimentation	1.6	1.5	1.4	1.2	1.6	1.6
Aerobic sedimentation	1.3	0.9	0.8	0.7	1.2	1.2
Diffusive flux	11	9.8	9.5	9.4	6.2	4-6
Internal loading	22	18	17	17	18	8-10
Settling flux	22	18	17	17	18	8-10
(settling-efflux)	-0.6	+0.4	+0.71	+0.8	+0.7	0.2-1
TP in water (ug/L)	45	38	36	36	40	25-35
SP in aerobic PW (mg/L)	0.13	.12	.11	.11	0.07	0.07
SP in anaerobic PW (mg/L)	1.9	1.9	1.9	1.9	1.9	1.9

PW = pore water
*Response after first year

Fig. 6.2a Soluble Phosphorus in Water Column for Year 3 of Null Option
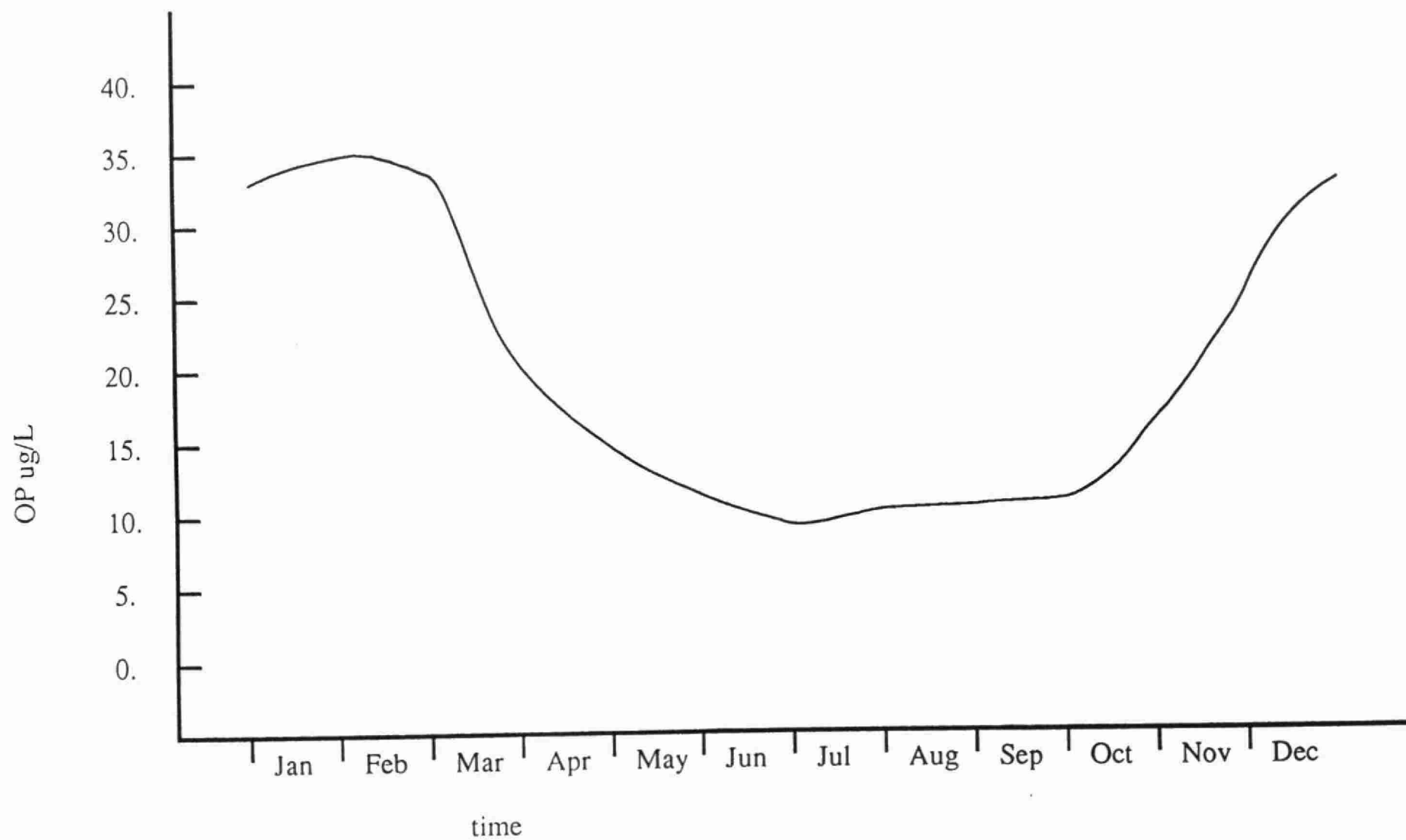
Fig. 6.2b Particulate Phosphorus in Water Column for Year 3 of Null Option
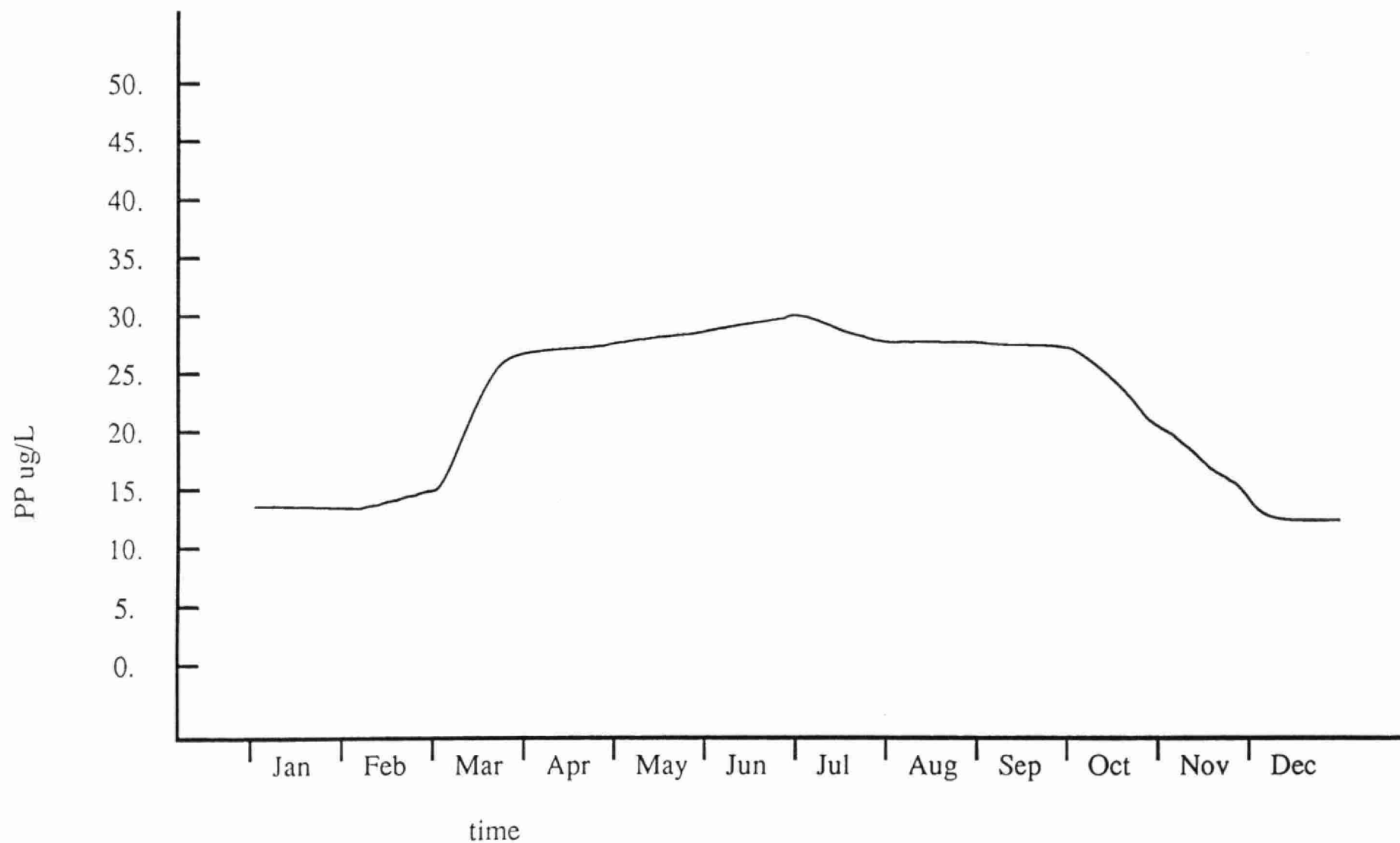
Fig. 6.2c Soluble Phosphorus in Aerobic Sediment for Year 3 of Null Option
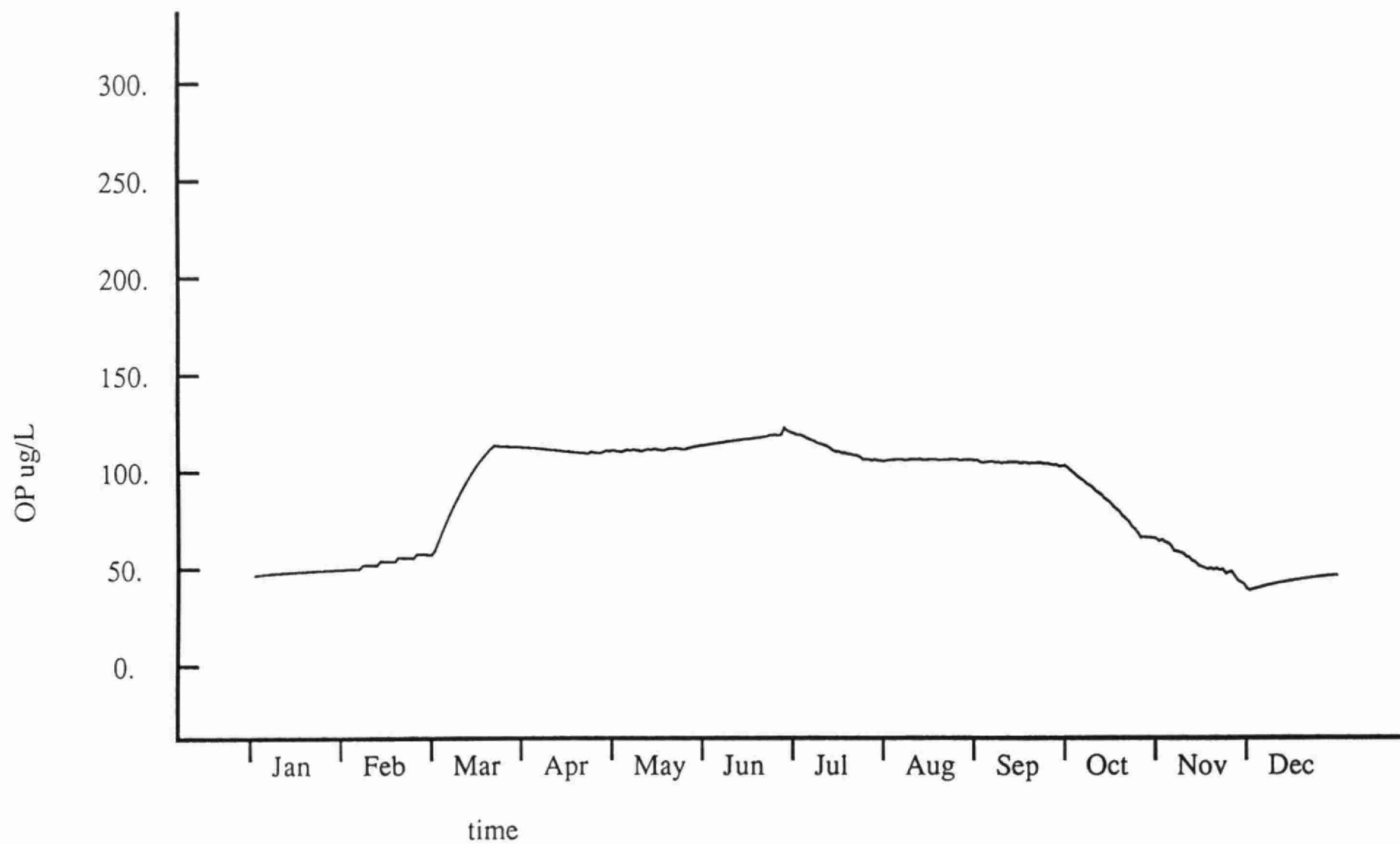
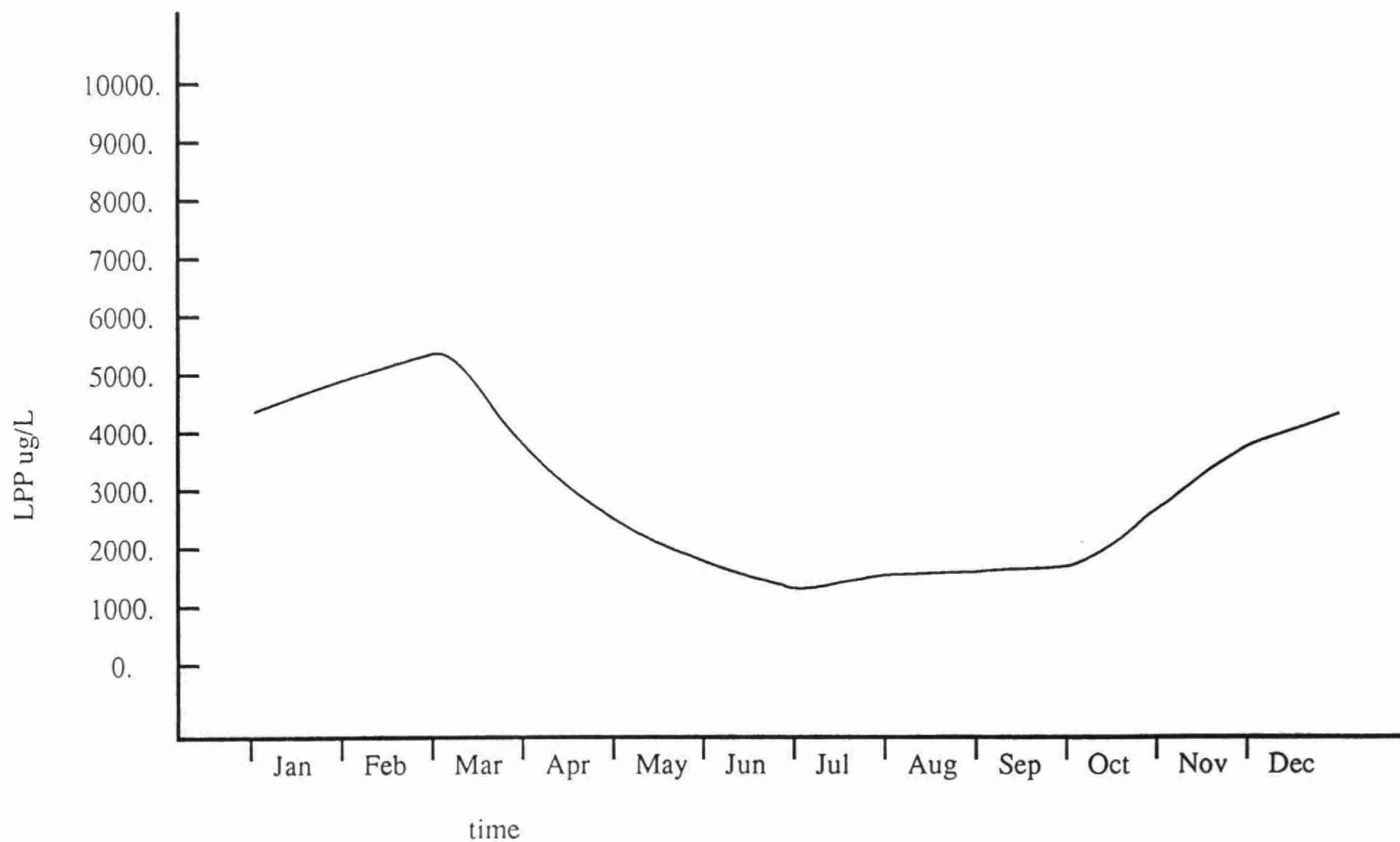Fig. 6.2d Labile Particulate Phosphorus in Aerobic Sediment for Year 3 of Null Option
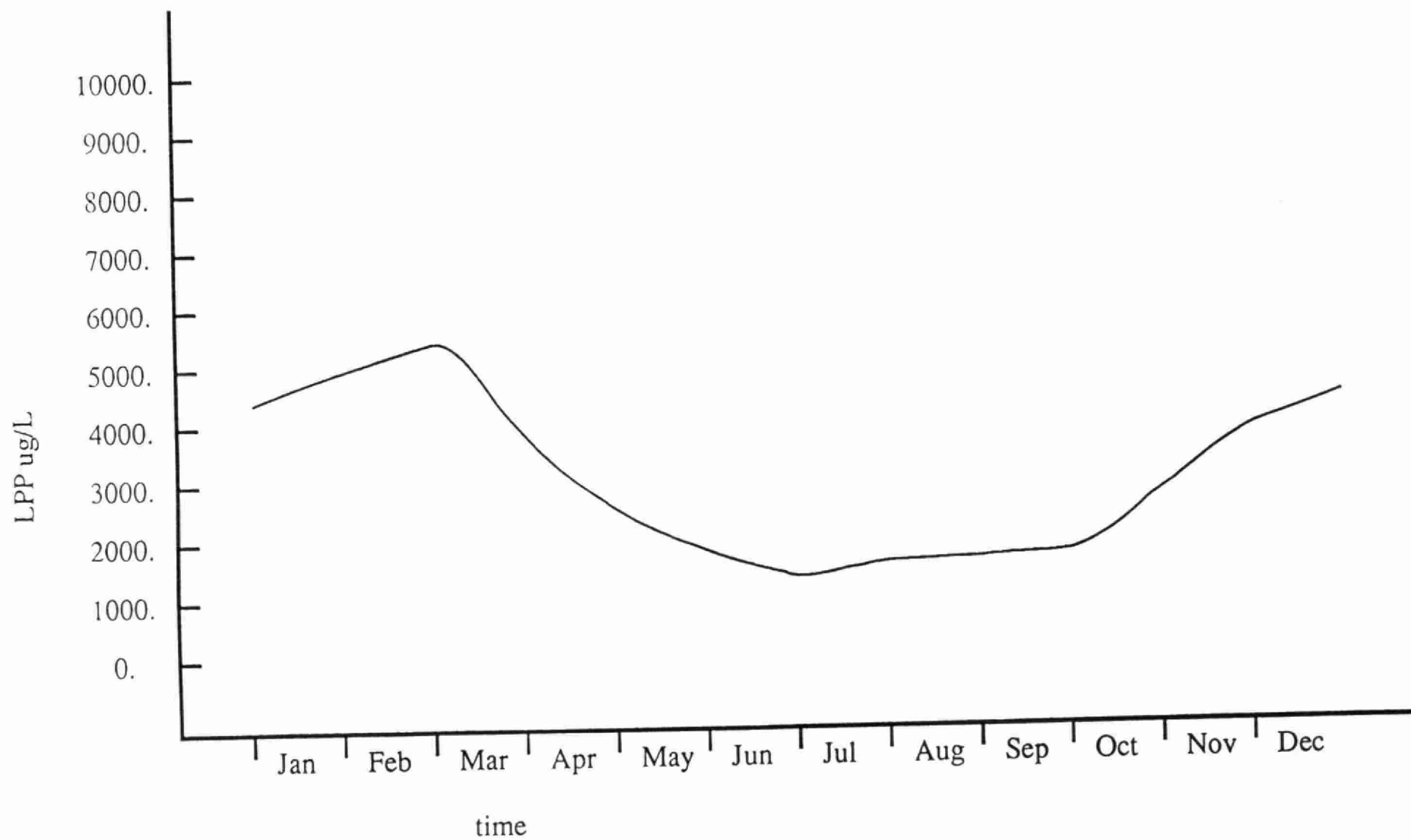
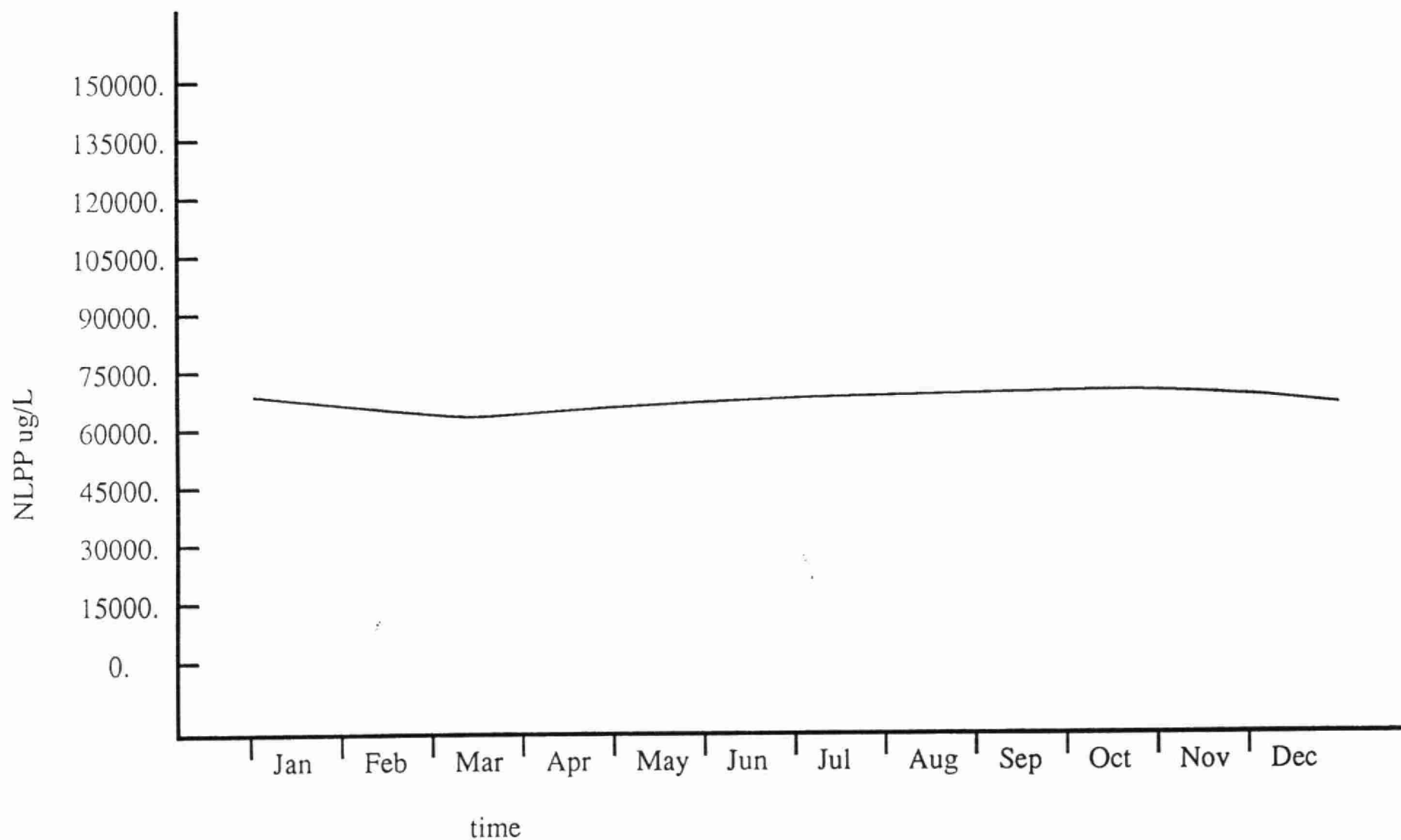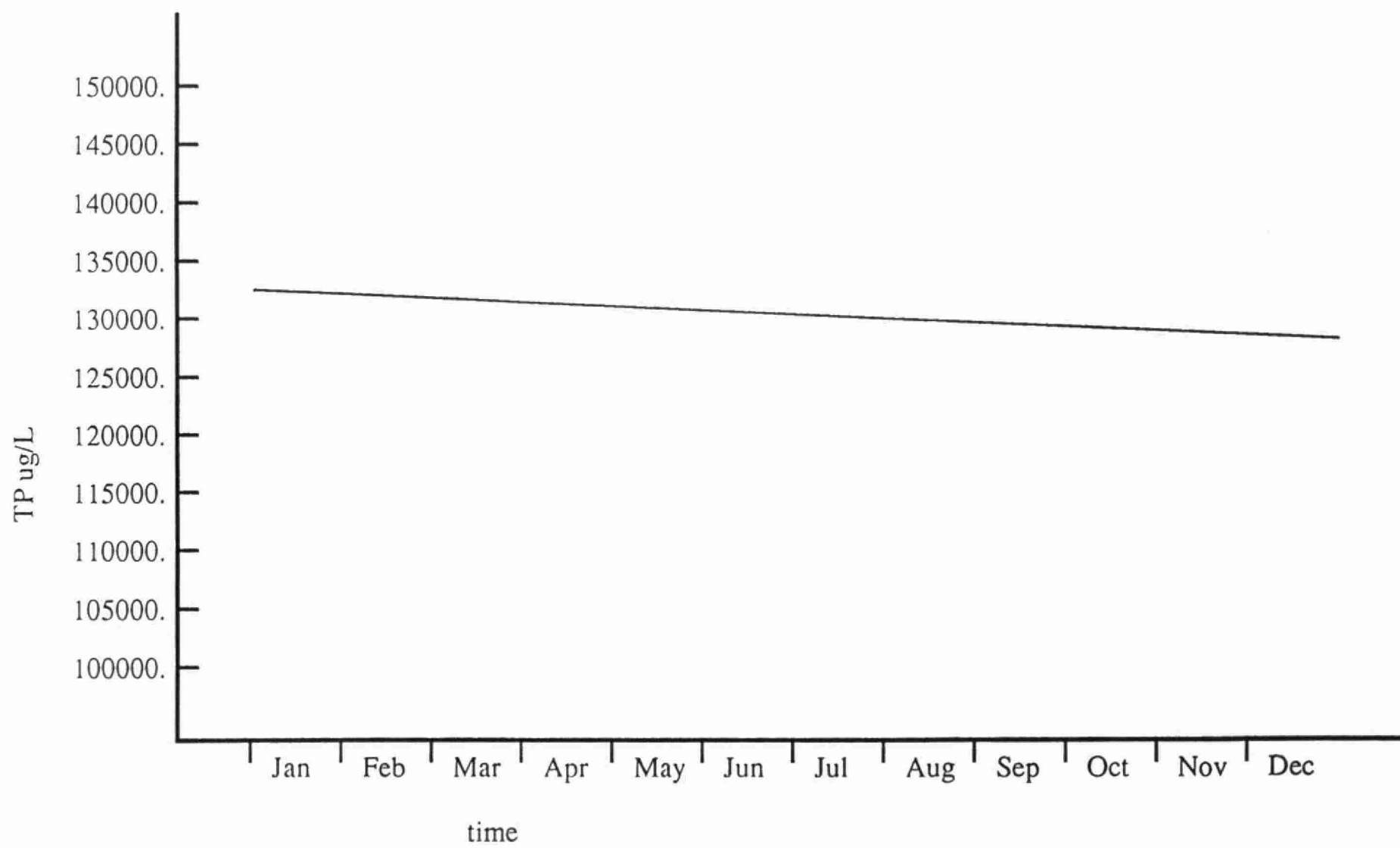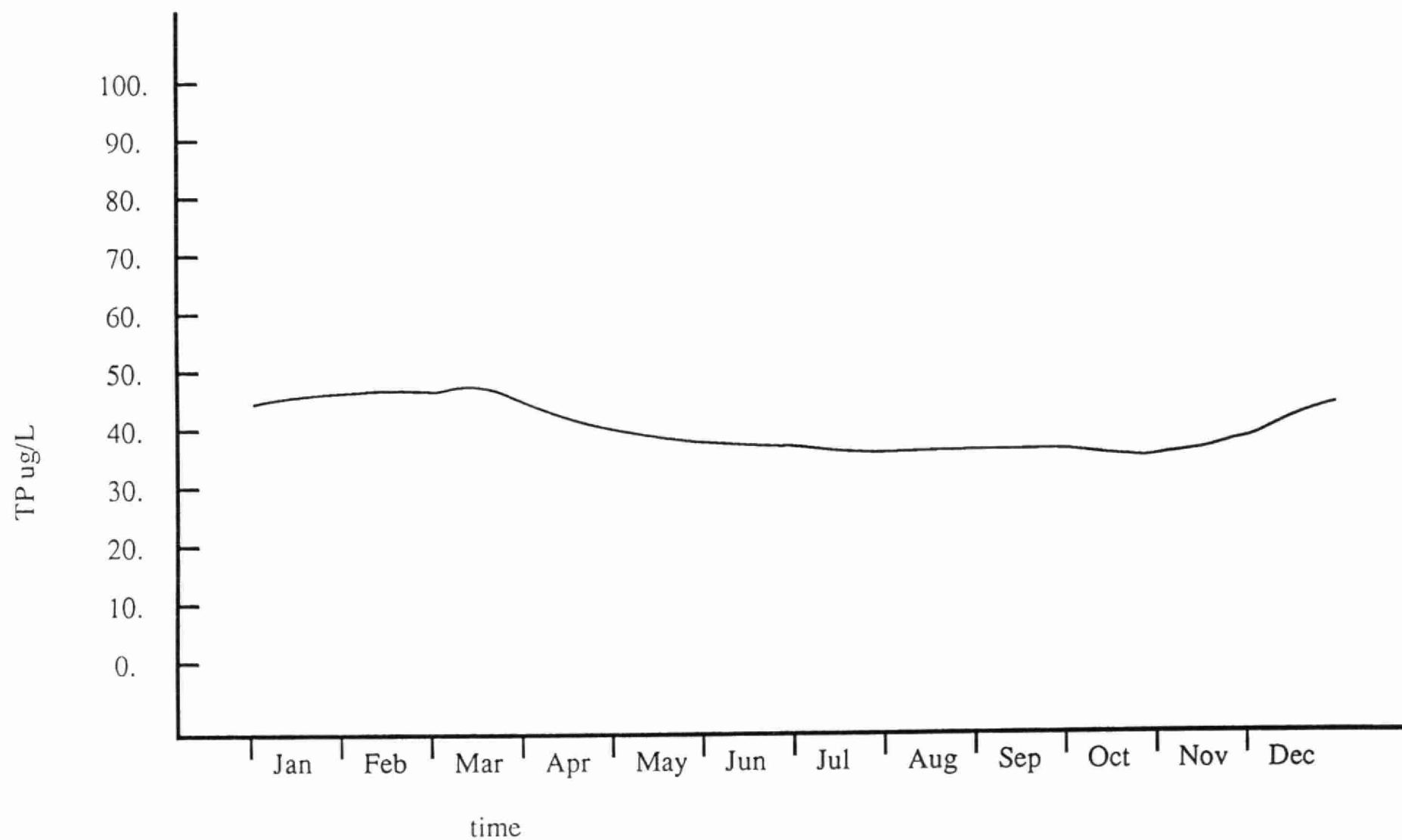Fig. 6.2e Non-labile Particulate Phosphorus in Aerobic Sediment for Year 3 of Null Option
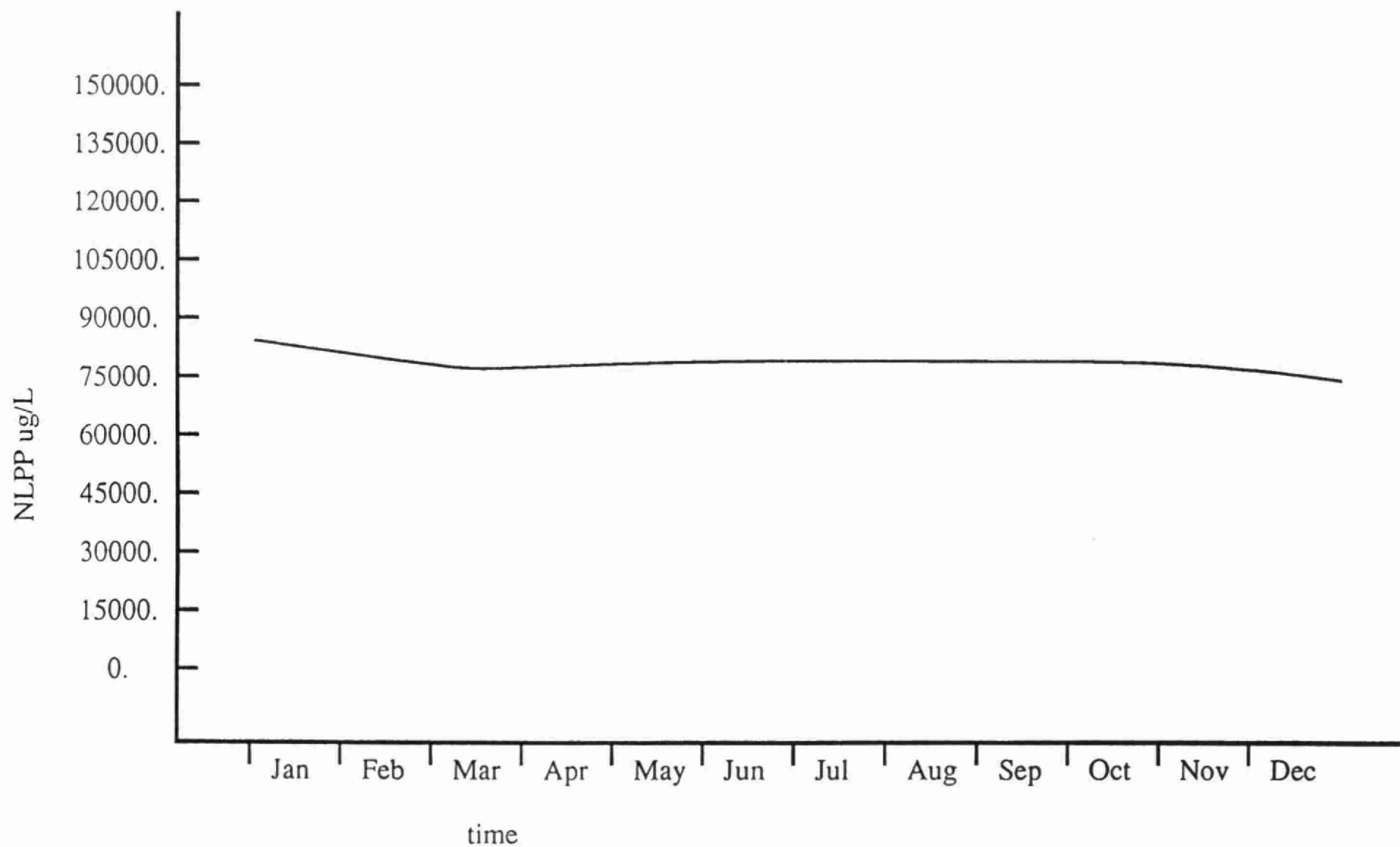
Fig. 6.2f Total Phosphorus in Anaerobic Sediment for Year 3 of Null Option
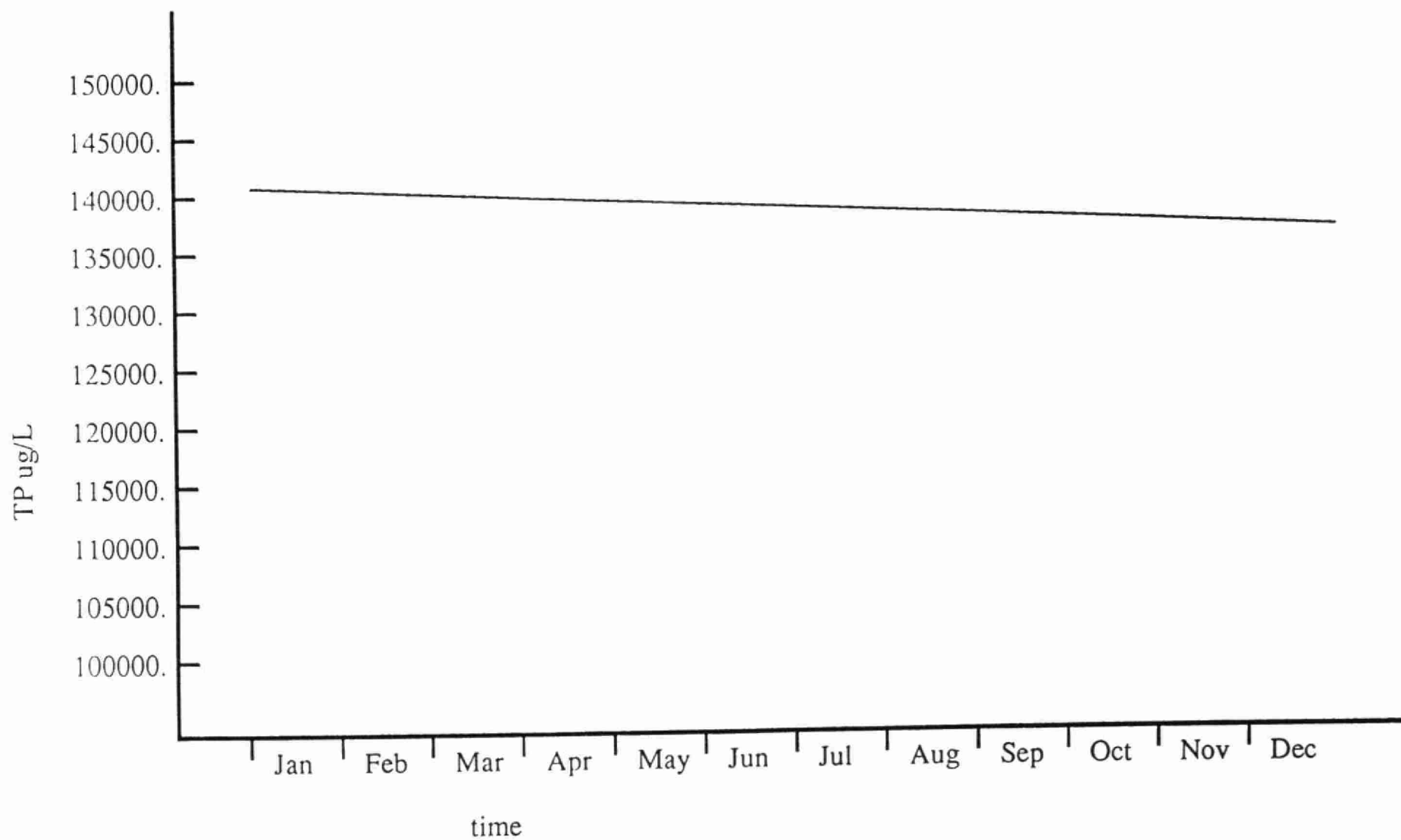
Fig. 6.2g Total Phosphorus in Water Column for Year 3 of Null Option
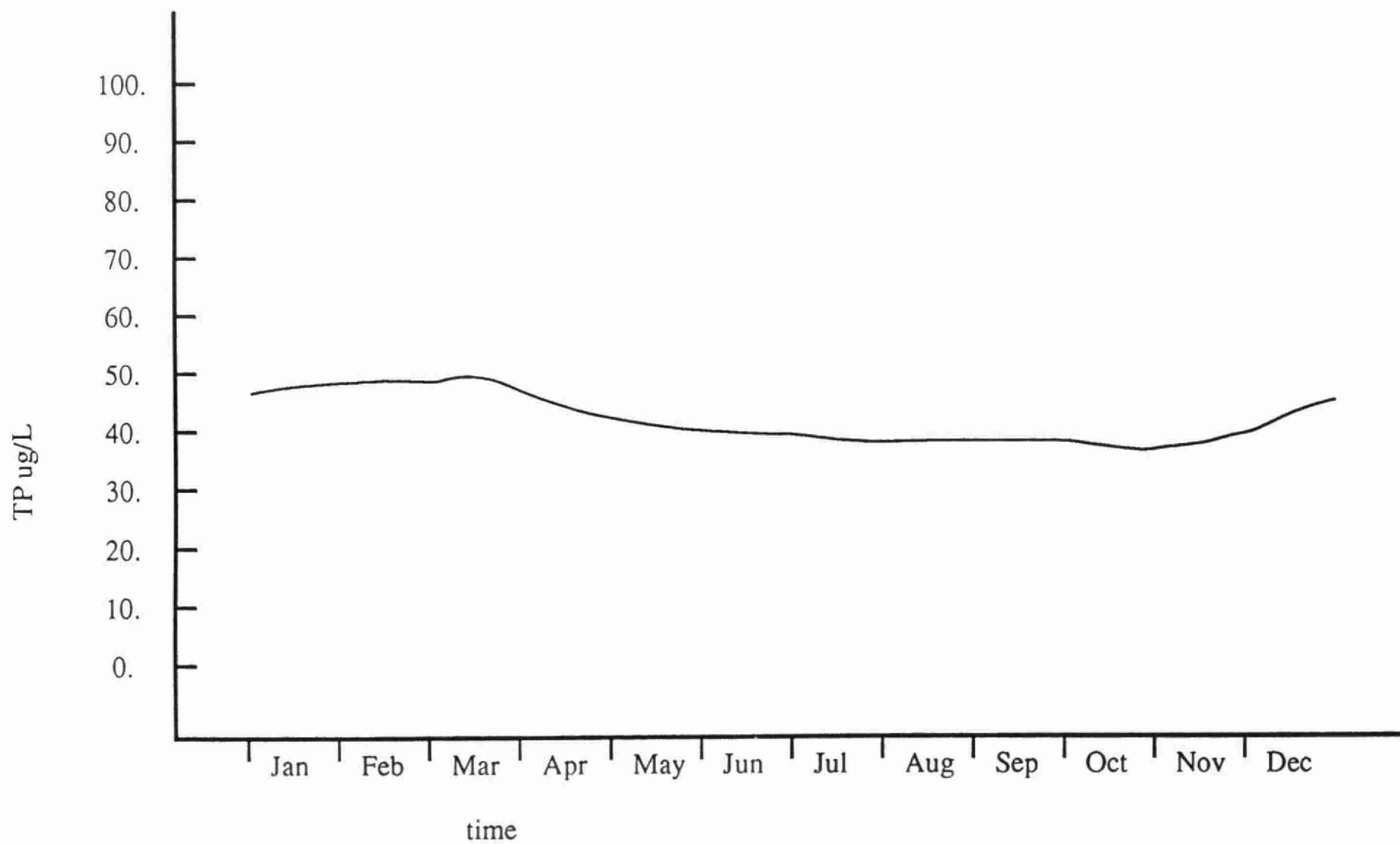
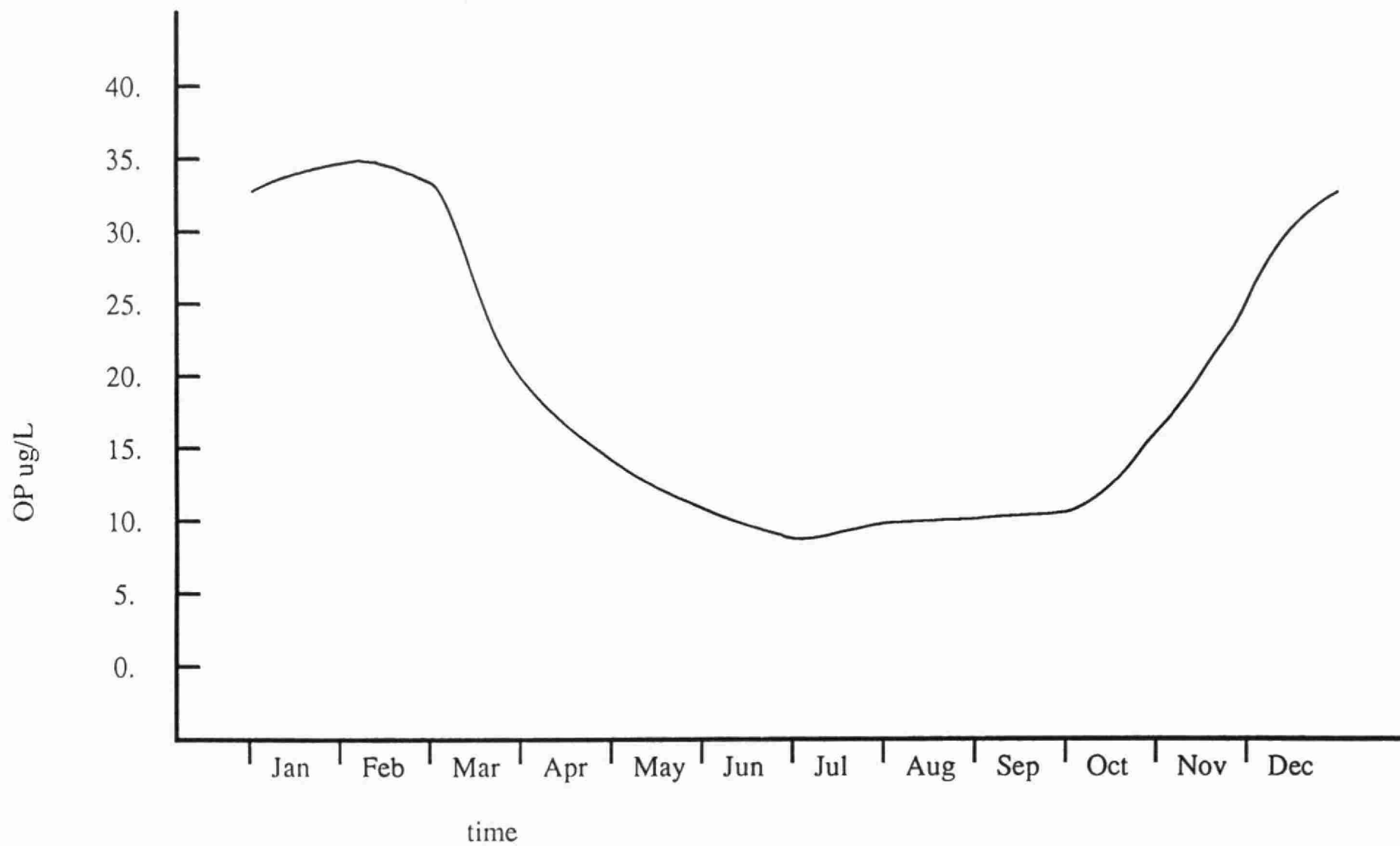Fig. 6.3a Soluble Phosphorus in Water Column for Year 5 of Null Option
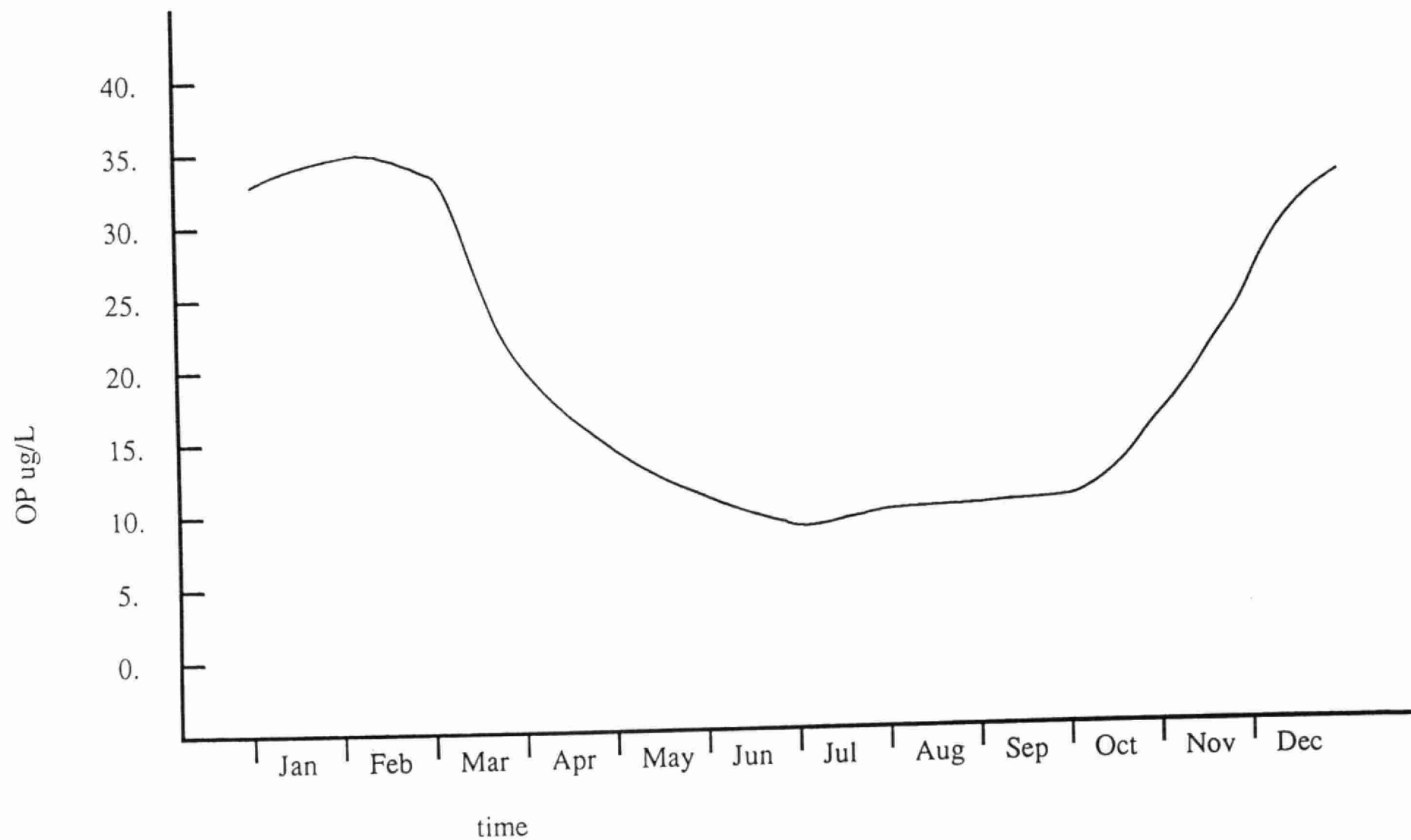
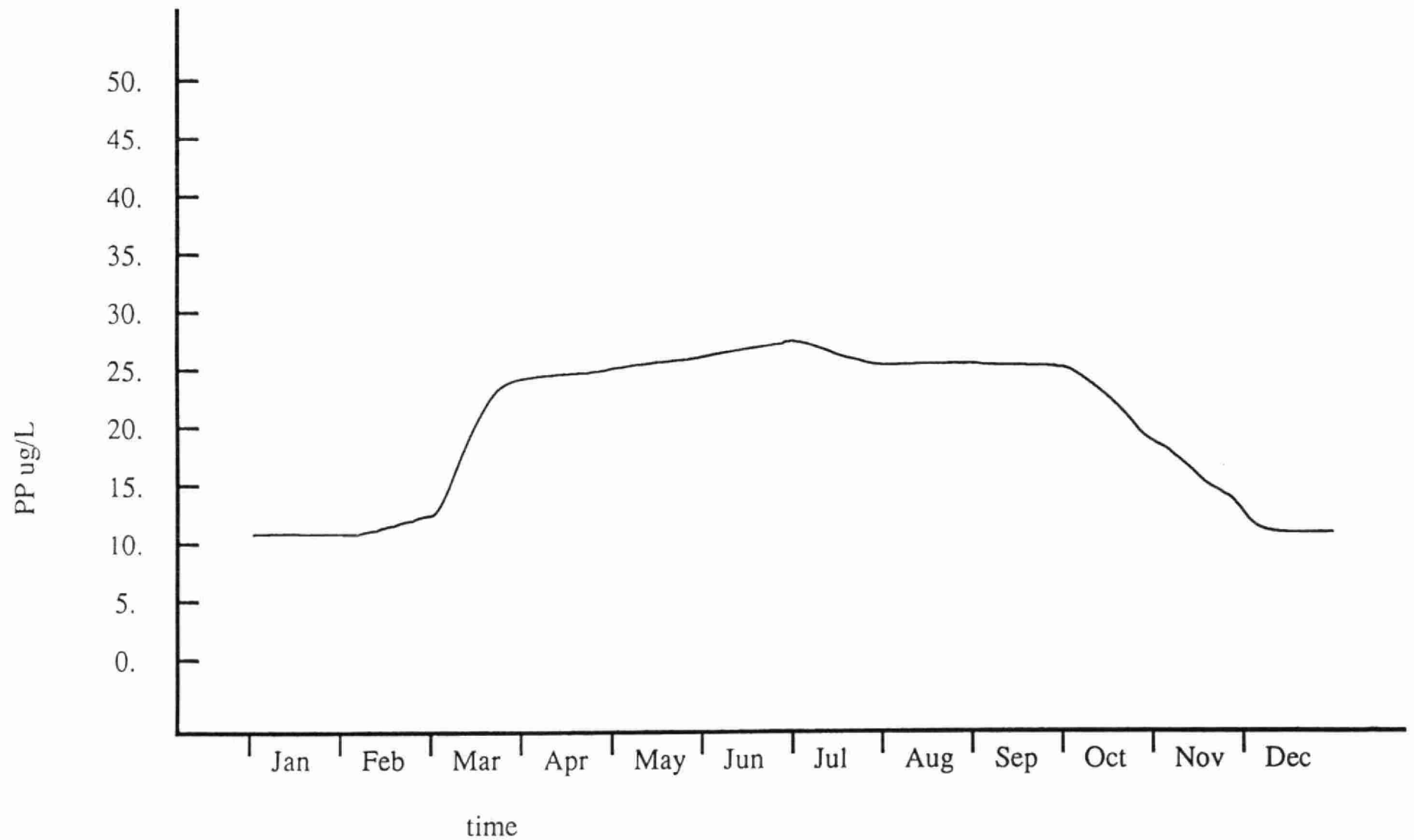Fig. 6.3b Particulate Phosphorus in Water Column for Year 5 of Null Option
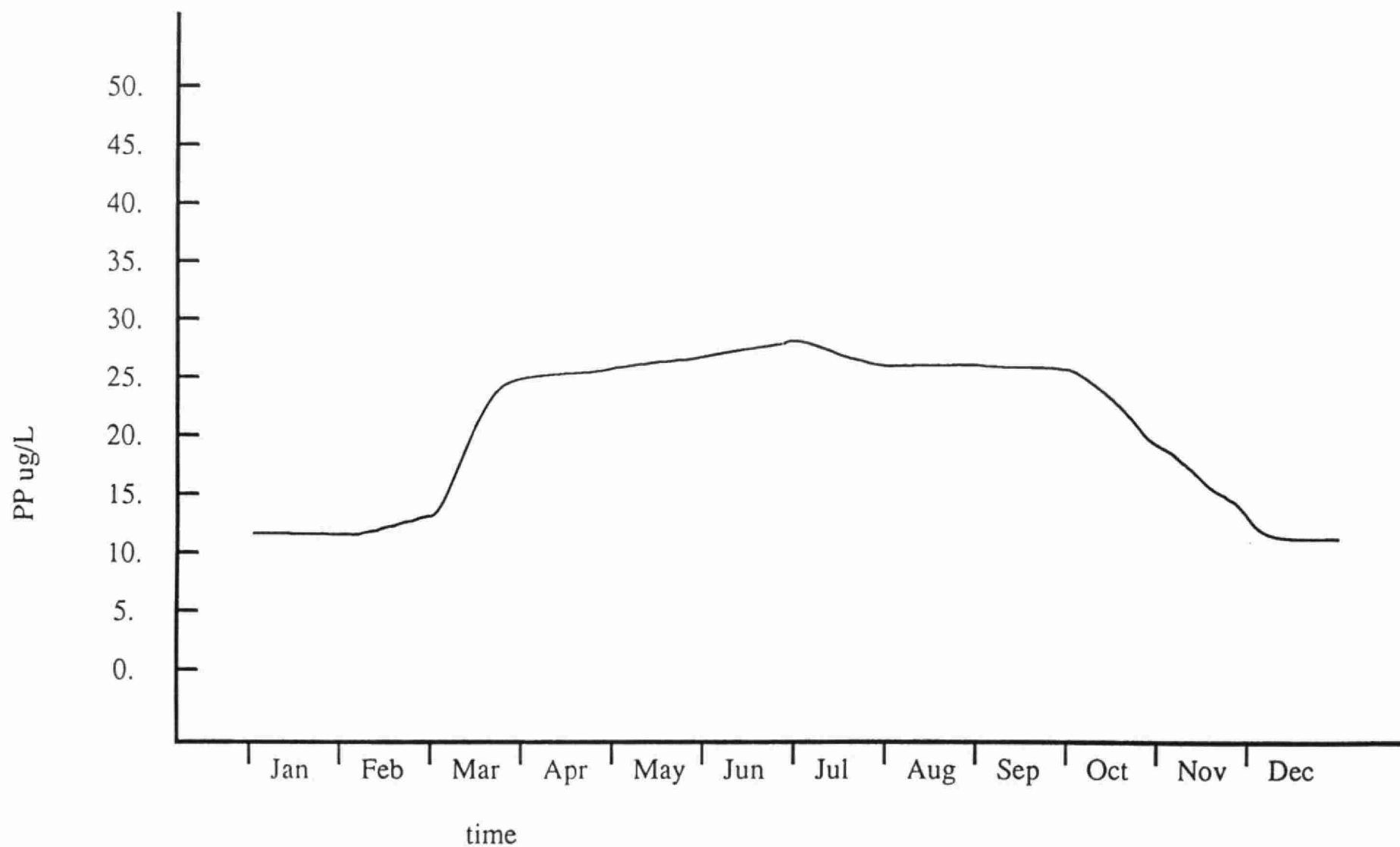
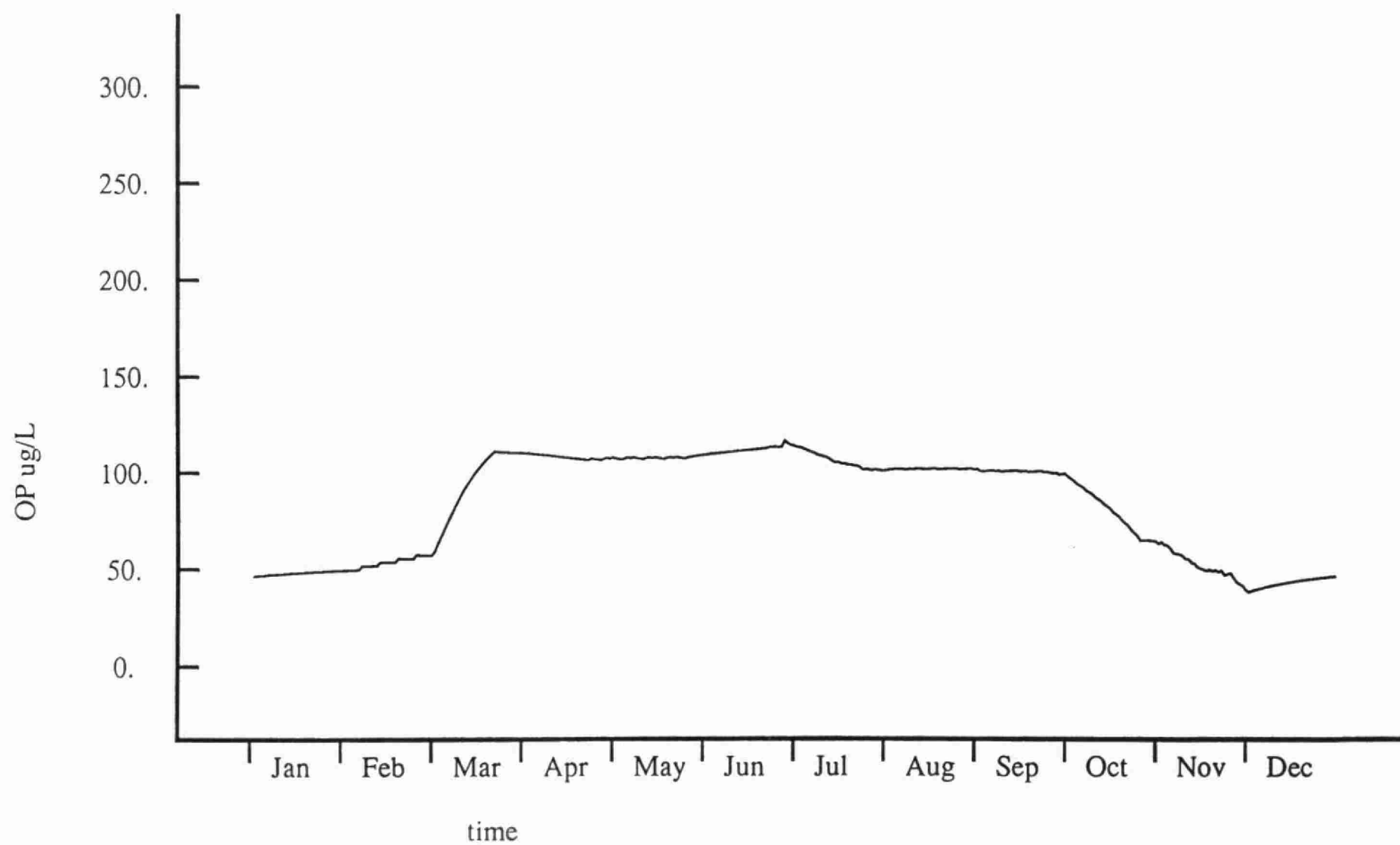Fig. 6.3c Soluble Phosphorus in Aerobic Sediment for Year 5 of Null Option
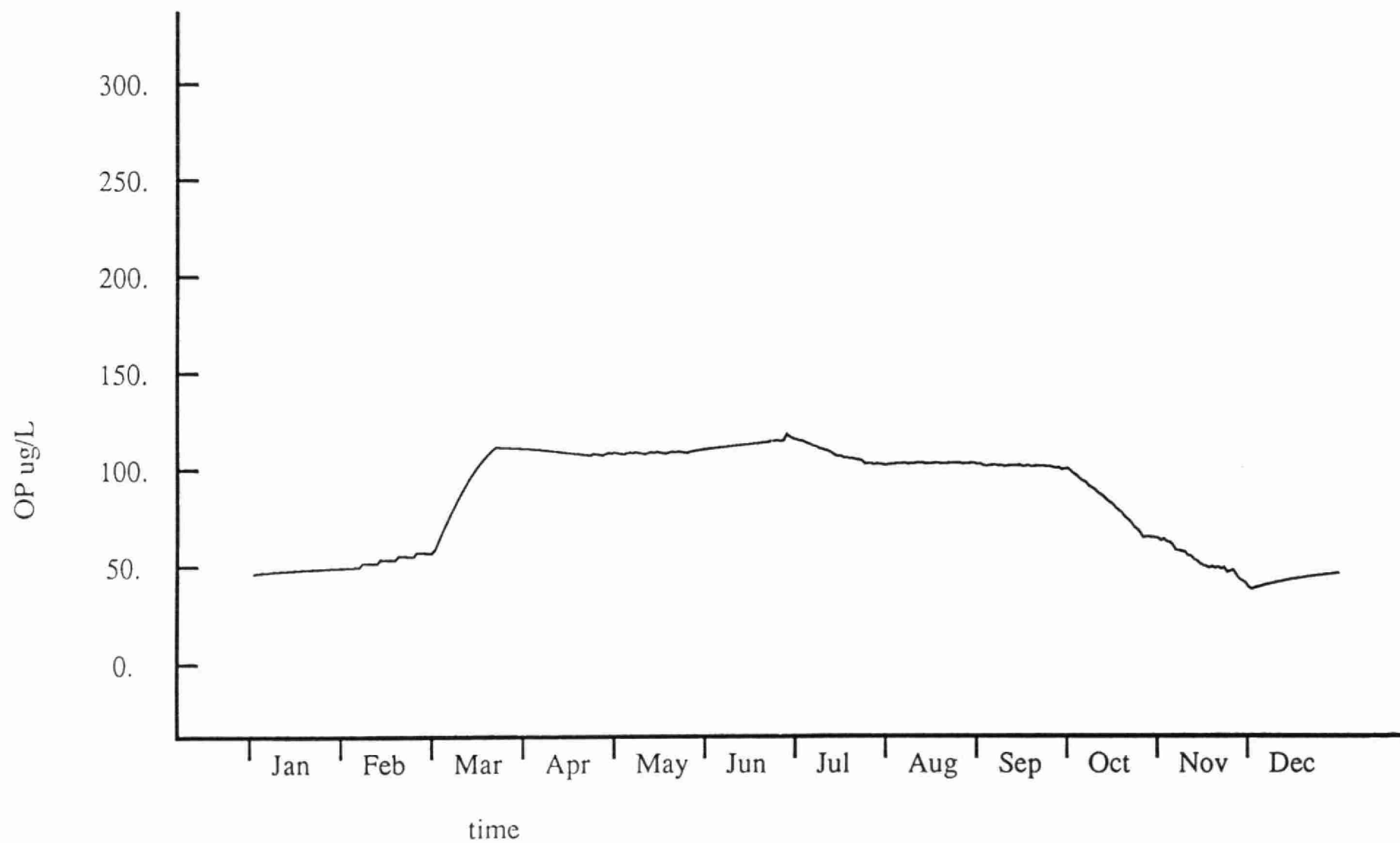
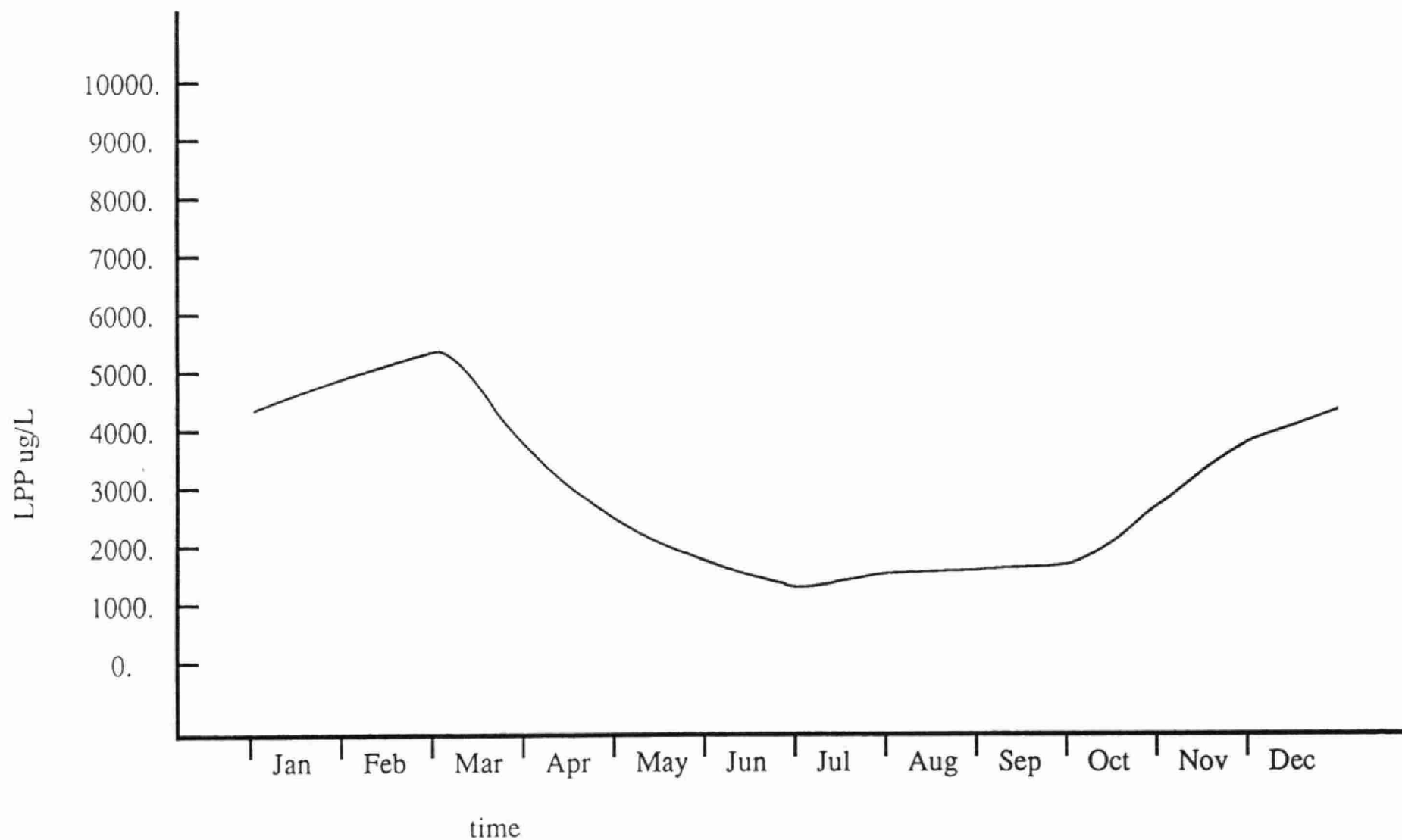Fig. 6.3d Labile Particulate Phosphorus in Aerobic Sediment for Year 5 of Null Option

Fig. 6.3e Non-labile Particulate Phosphorus in Aerobic Sediment for Year 5 of Null Option

Fig. 6.3f Total Phosphorus in Anaerobic Sediment for Year 5 of Null Option

Fig. 6.3g Total Phosphorus in Water Column for Year 5 of Null Option

Fig. 6.4a Soluble Phosphorus in Water Column for Year 10 of Null Option

Fig. 6.4b Particulate Phosphorus in Water Column for Year 10 of Null Option

Fig. 6.4c Soluble Phosphorus in Aerobic Sediment for Year 10 of Null Option

Fig. 6.4d Labile Particulate Phosphorus in Aerobic Sediment for Year 10 of Null Option

Fig. 6.4e Non-labile Particulate Phosphorus in Aerobic Sediment for Year 10 of Null Option
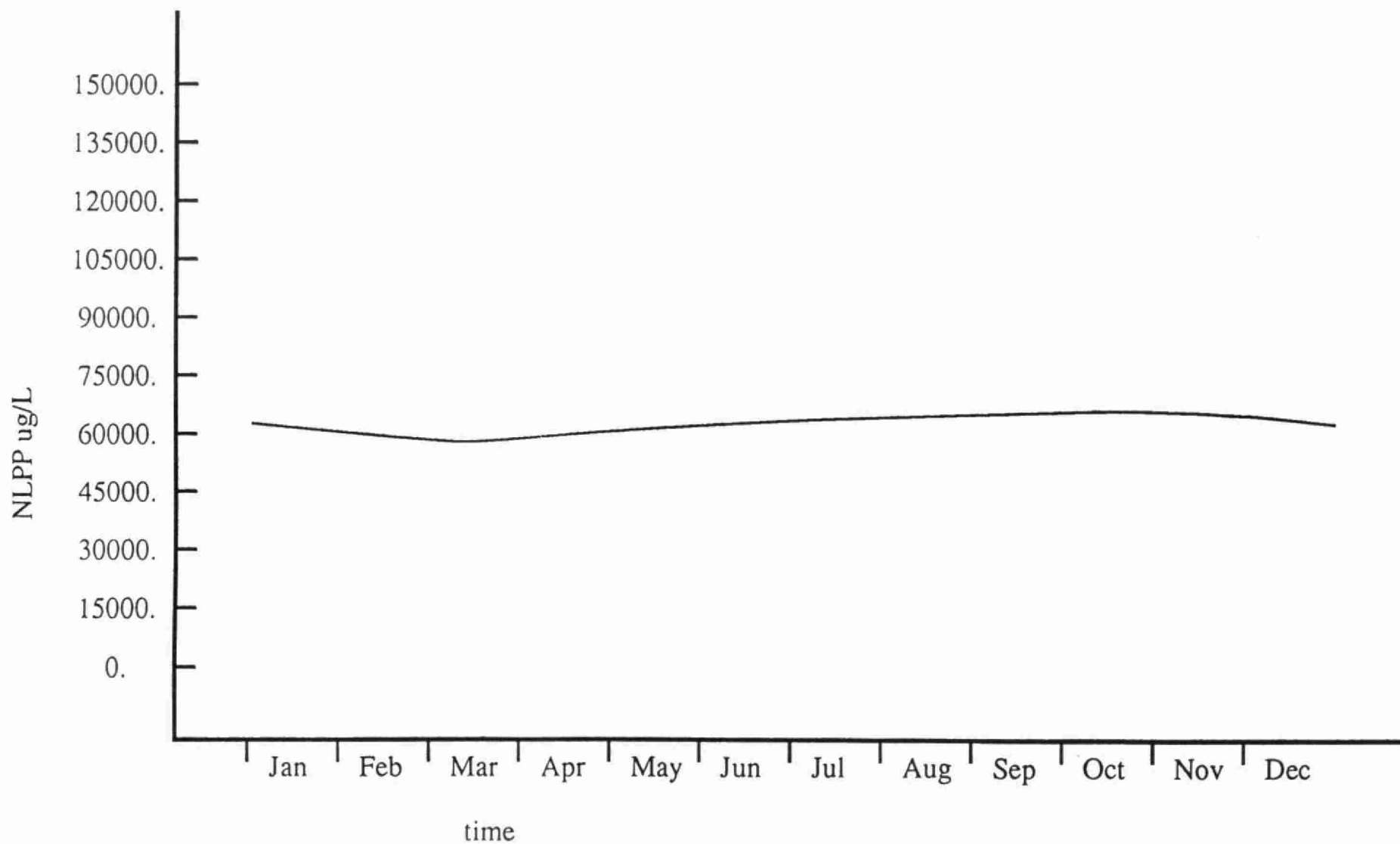
Fig. 6.4f Total Phosphorus in Anaerobic Sediment for Year 10 of Null Option
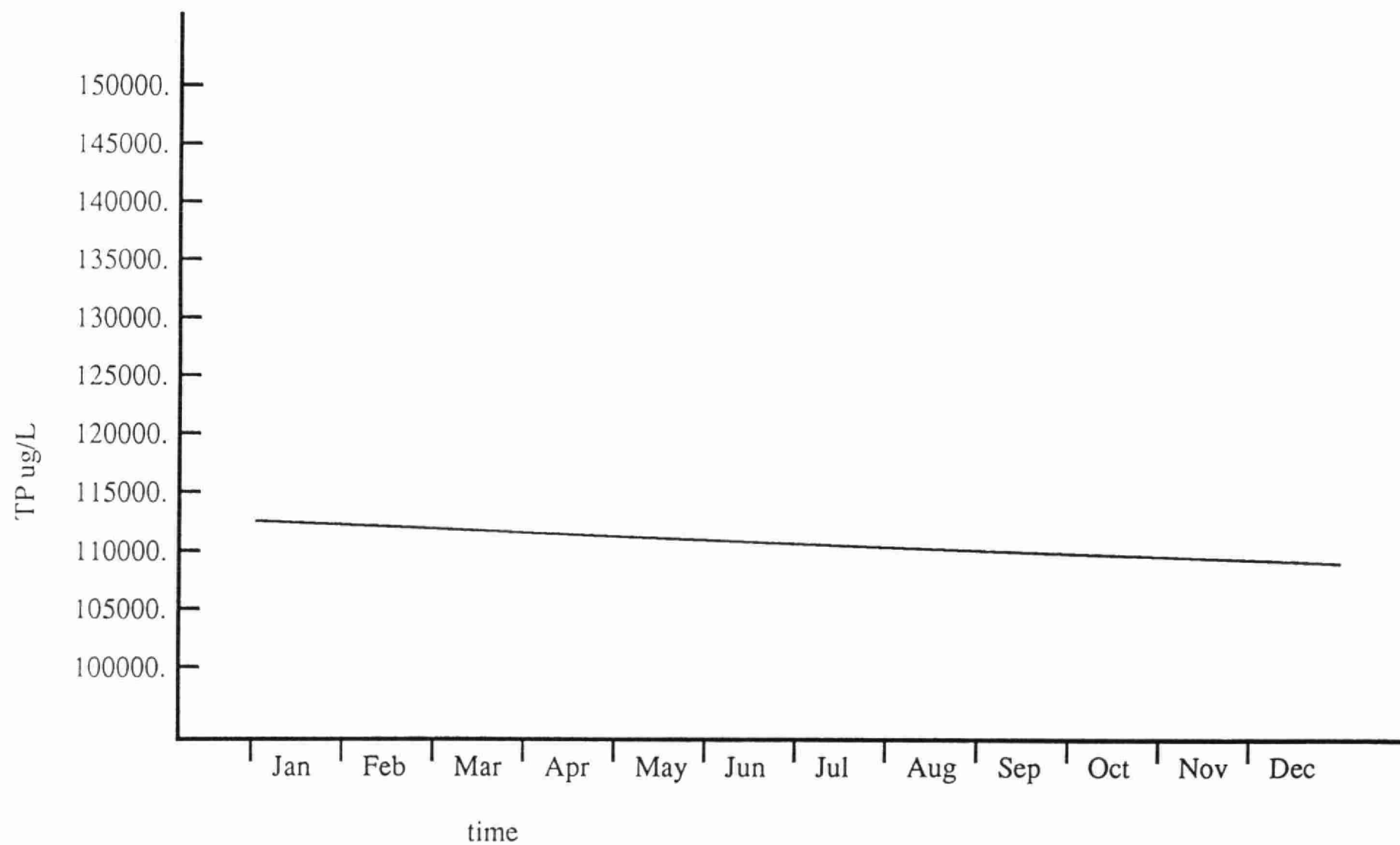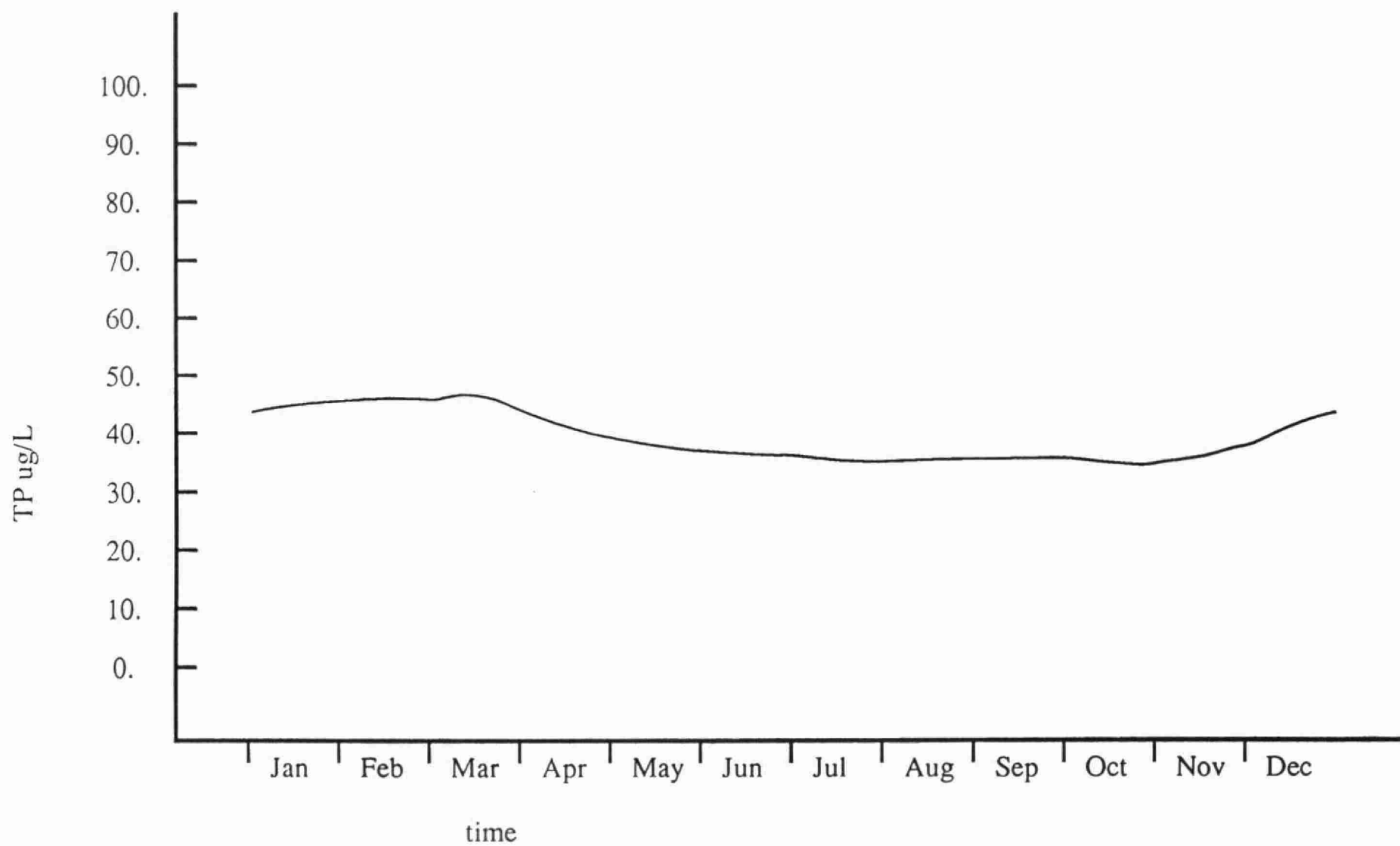
Fig. 6.4g Total Phosphorus in Water Column for Year 10 of Null Option

The calculated response in fluxes and concentrations for the null option and for an aluminum treatment are given in Table 6.3. Overall, they show that the deep sediments continue to clean themselves up over a decade or more, but that the surficial sediments have generally responded in less than 5 years. Furthermore, despite the initial efflux being larger than the initial settling flux, the settling flux is larger than the efflux within two years. Finally, a new steady state concentration is attained within three years of initial conditions with a water column concentration of 36-38 ug/L. This decrease is consistent with observed data in which concentrations in the Upper Bay decreased from total P levels of 50 ug/L in 1984-85 to the 33-37 ug/L (euphotic zone) range in 1986 and 1987 (C. Minns, personal communication).

6.7.2   Effectiveness of Alum Injection

Two evaluations of the effectiveness of alum injection are indicated in Table 6.3: one assuming that aluminum only affects the pore water concentration of phosphorus; and one assuming that it affects the sediment resistance to erosion. The effect of alum injection, if it only affects the pore water concentration of phosphorus, is given in Figure 6.5. The effect of aluminum control upon phosphorus in pore water is much more immediate than the effects of natural restoration. It reduces the diffusive flux to a minimal amount. However, the overall reduction in total phosphorus in the water column is less than the calculated reduction for the null option, and, based upon literature, would be needed to be re-applied every two years.

Some care must be used in assessing the effectiveness of aluminum upon the diffusional flux of phosphorus since there are some uncertainties resulting from the experimentation above which need to be measured before this conclusion can be taken as definitive. One major concern is whether the lab work on which this assessment was based was truly dosed correctly.

A more major concern is the physical effect of alum as a coagulant. As a coagulant, it would increase the ability of the sediments to resist erosion by increasing their shear strength. This may have a larger effect than chemical equilibrium. Testing is required to evaluate this possibility, especially since a substantial portion of the phosphorus return is generated by erosion of sediment particles. A speculative impact of alum control upon particulate erosion is also indicated in Table 6.3. It assumes that approximately 50% of

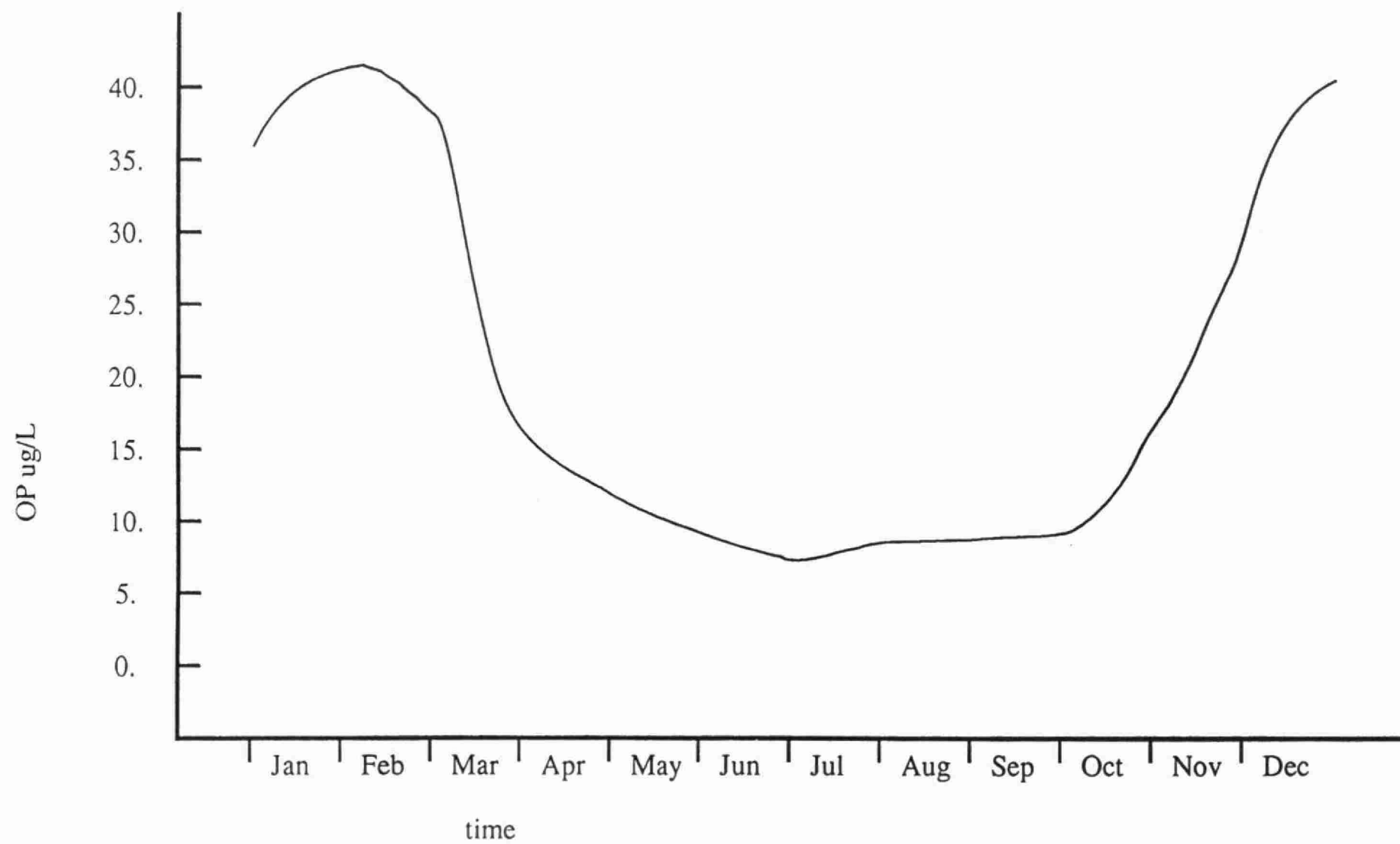Fig. 6.5a Soluble Phosphorus in Water Column for Alum Control

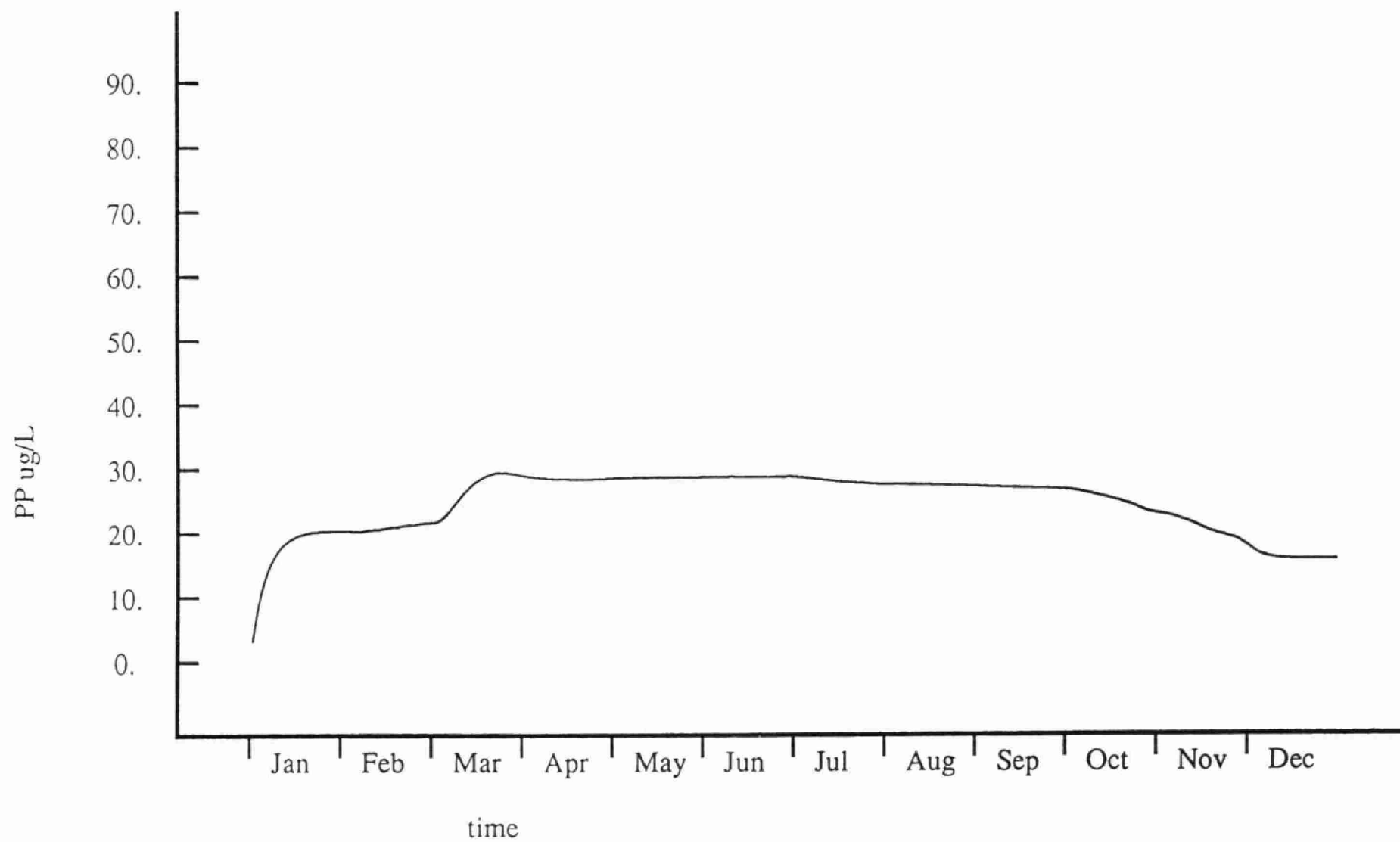Fig. 6.5b Particulate Phosphorus in Water Column for Alum Control

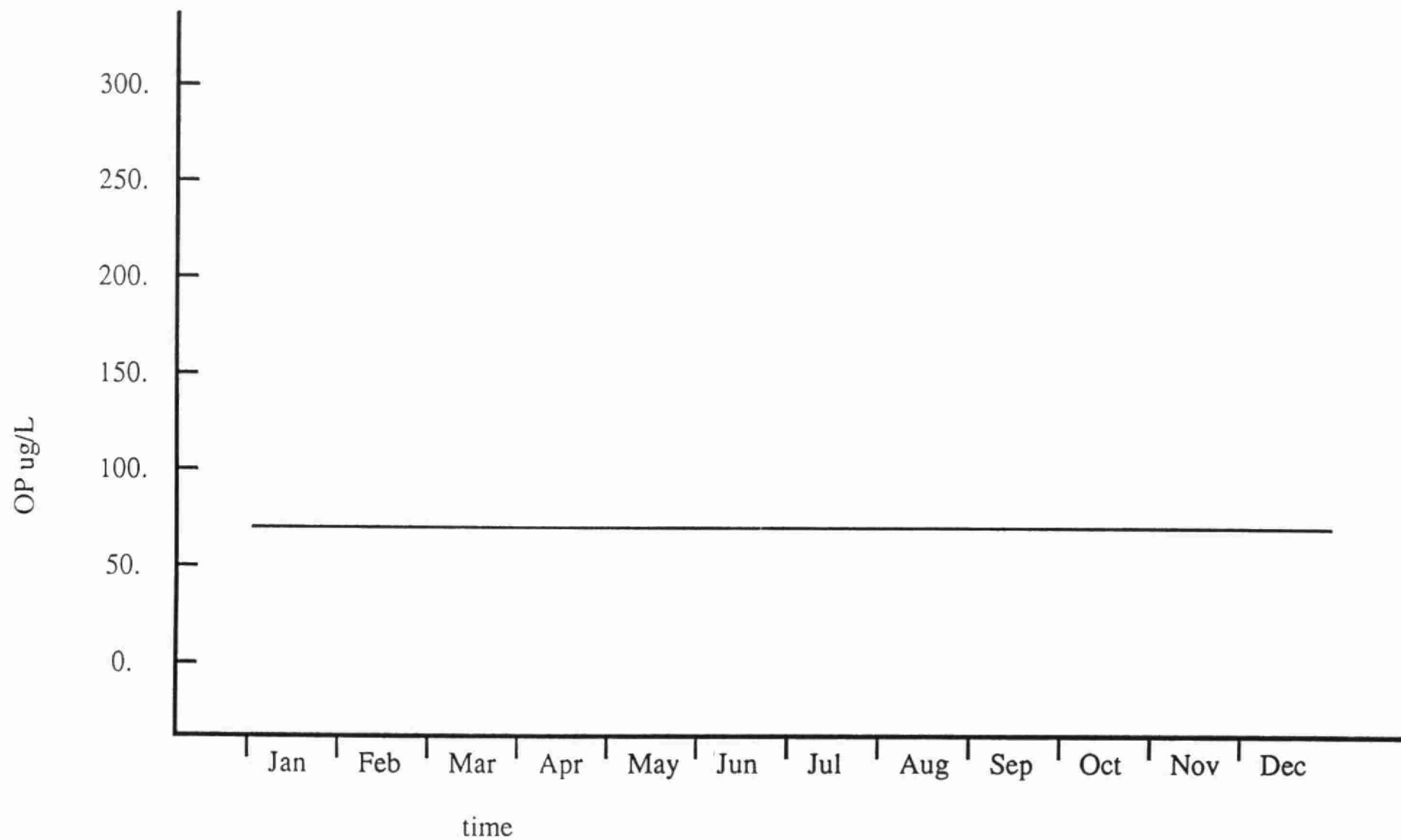Fig. 6.5c Soluble Phosphorus in Aerobic Sediment for Alum Control

Fig. 6.5d Labile Particulate Phosphorus in Aerobic Sediment for Alum Control
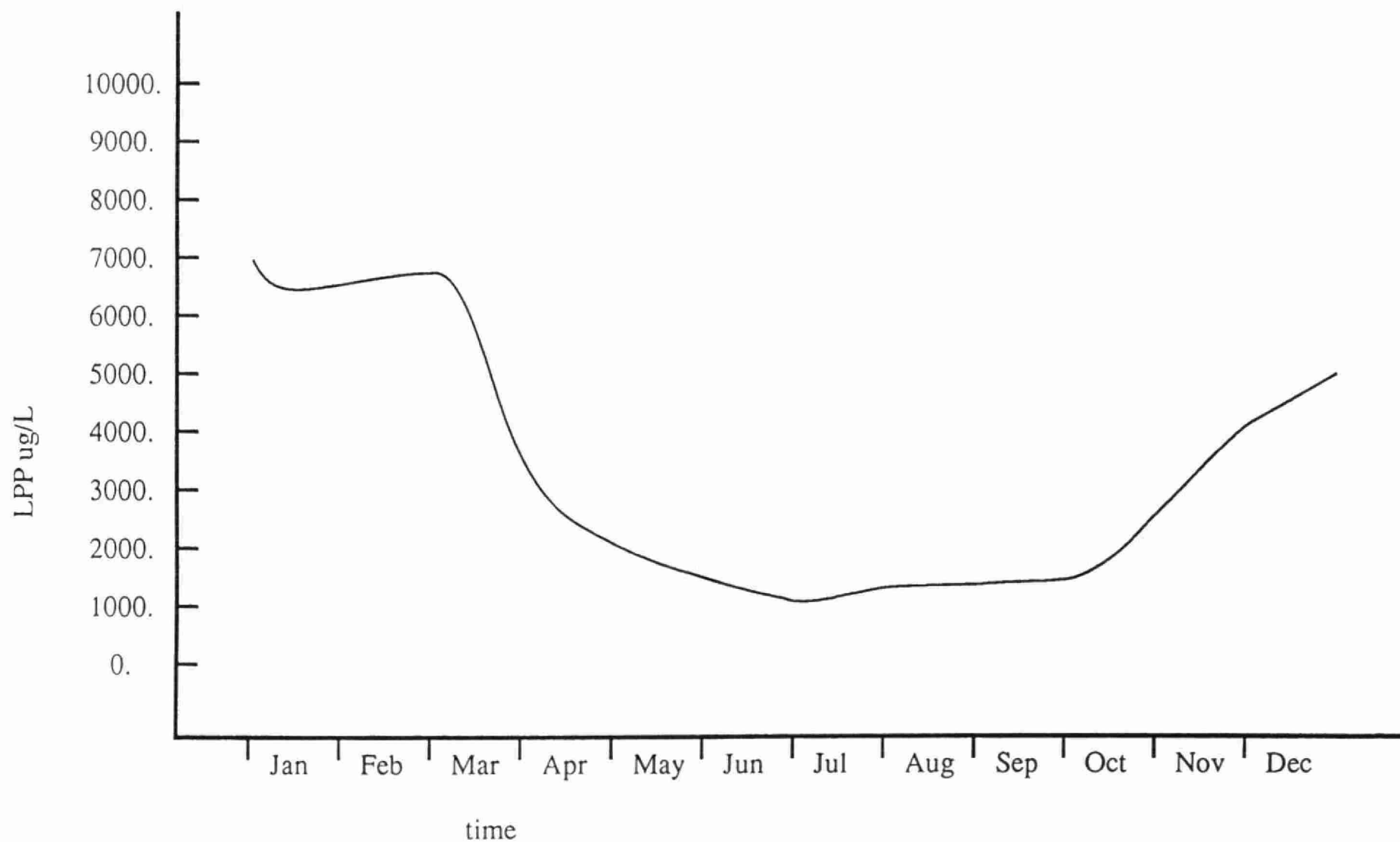
Fig. 6.5e Non-labile Particulate Phosphorus in Aerobic Sediment for Alum Control
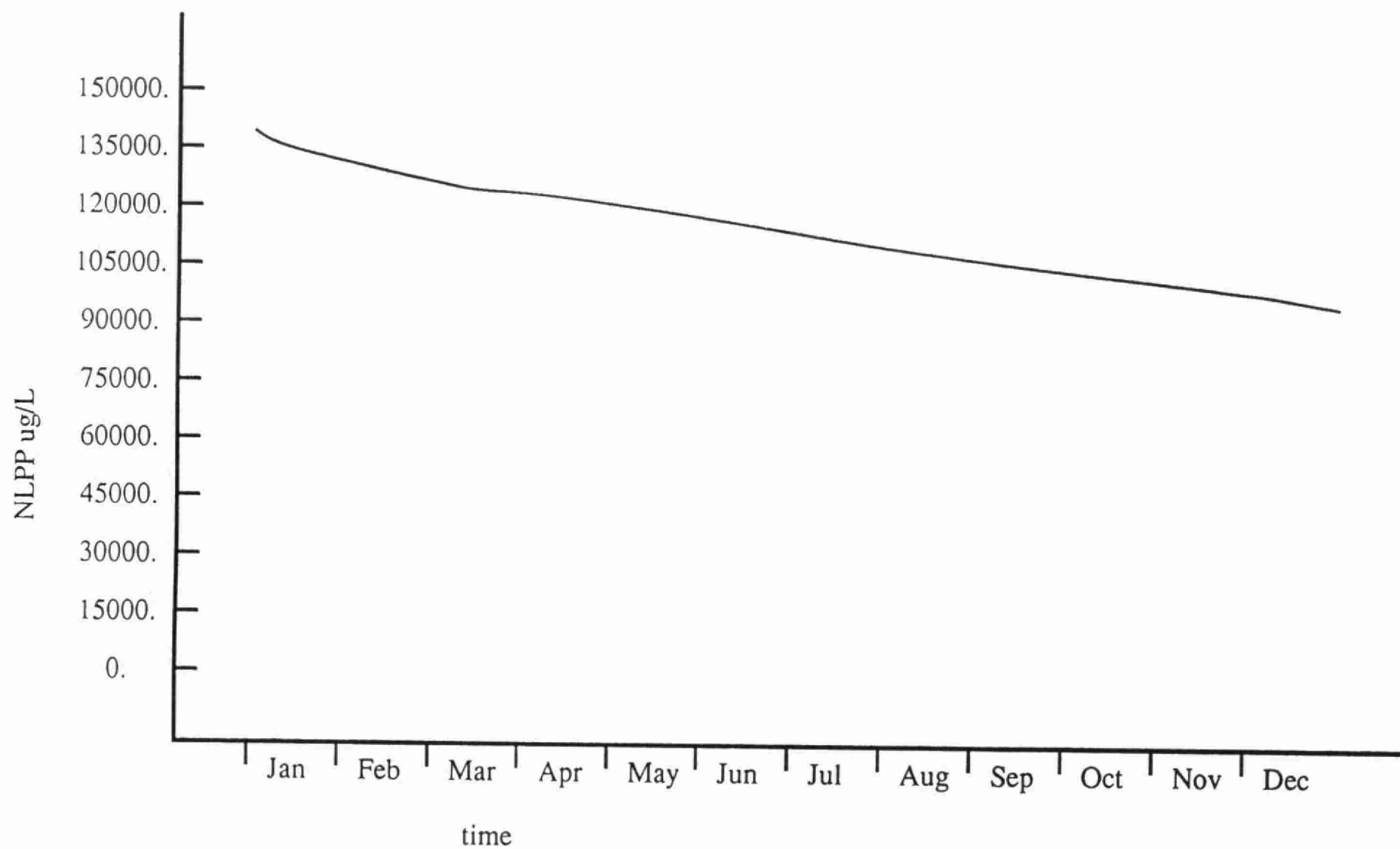
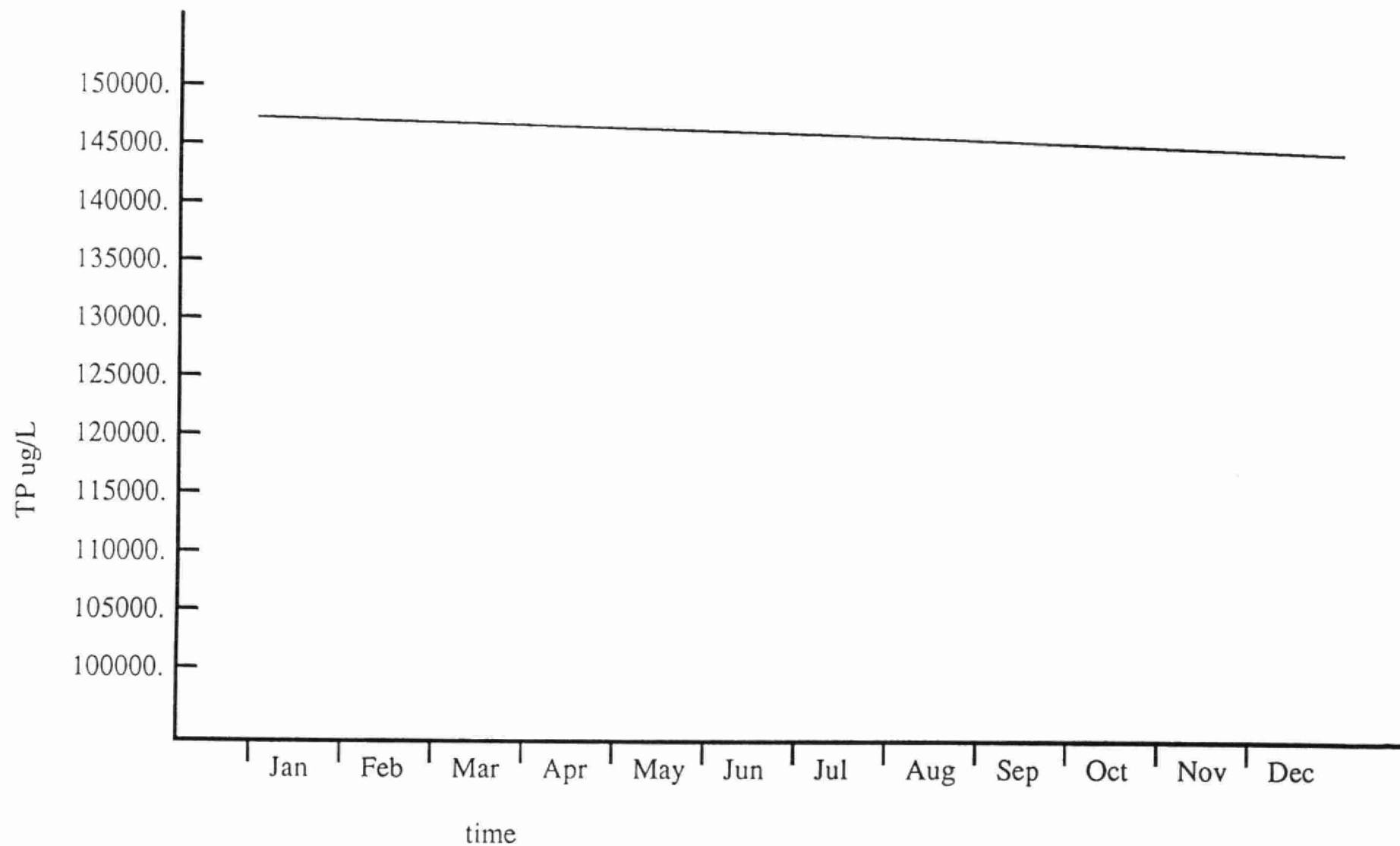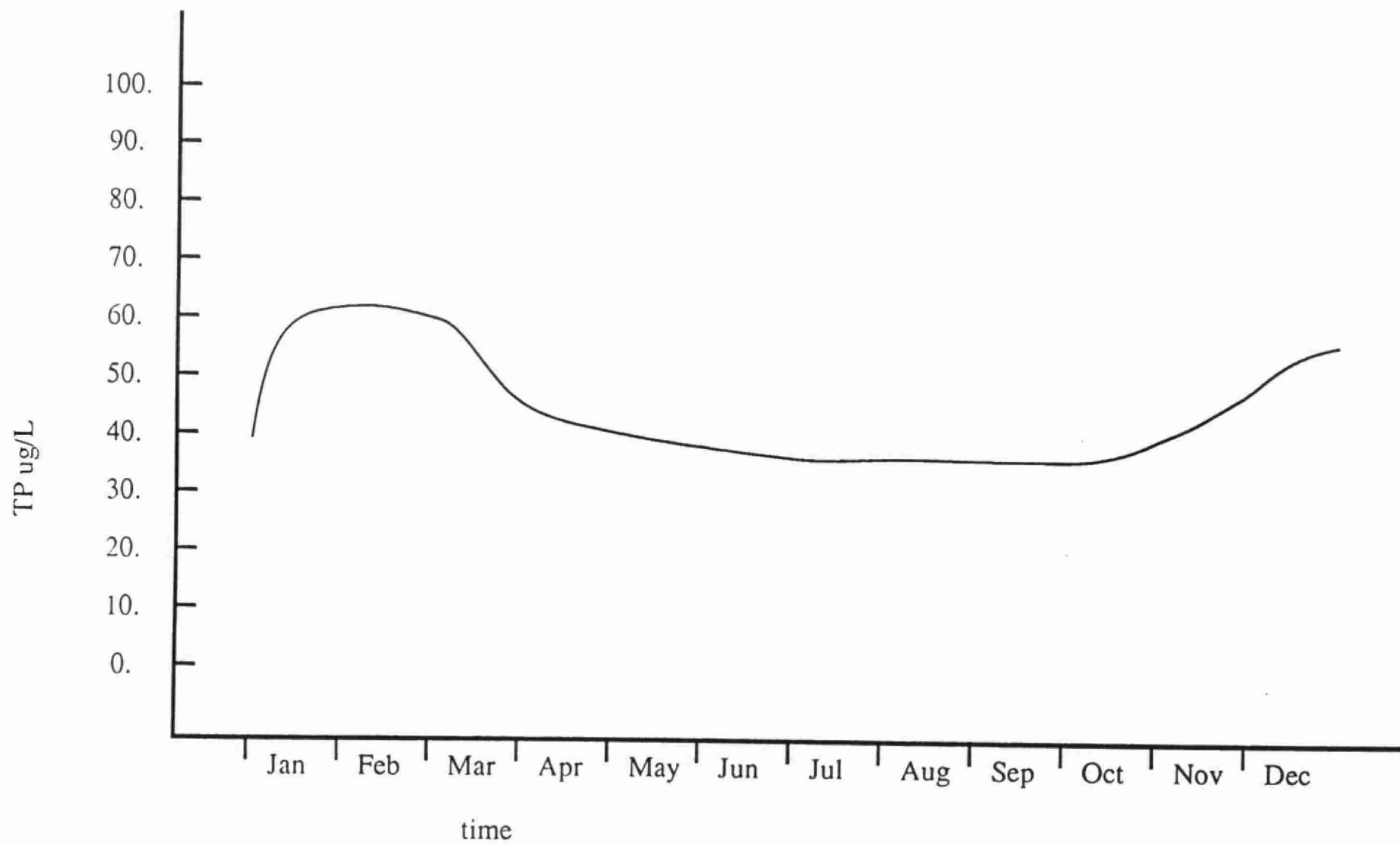Fig. 6.5f Total Phosphorus in Anaerobic Sediment for Alum Control

Fig. 6.5g Total Phosphorus in Water Column for Alum Control

the internal loading from erosive action is controlled by increasing the shear strength of the sediments.

6.8   Overall Assessment of Modelling and Experimental Results

The modelling assessment indicates that a plausible set of coefficient values have been obtained which give reasonable agreement between predicted values and various environmental observations.   Due to the near similarity of values calculated for reduction in internal loading for the null option and the effect of alum addition, further modelling, field and laboratory studies are in order to improve our confidence in the calculated results.   In particular, studies which differentiate chemical effects from physical effects are required.

In the near short term, additional modelling studies should concentrate upon validation of the present model's calculations with field data available for a 15 year period or so and be extended to describe fluxes of SRP from deeper sediments.   Some additional inspection of the geochemical interactions is required.   Also, in the short term (this field year if possible), it would be useful to get a set of proper data for SRP and several metals not previously measured to evaluate the geochemical assumptions involved.

Over a slightly longer time scale, it is in order to repeat some of the phosphorus precipitation experiments with alum injection to better define its effect upon pore water levels and sediment structure.

Over the long term, additional phosphorus flux measurements and a full scale demonstration of the impact of alum addition would be useful.

6.9   Further Studies

Further studies are required to increase our understanding of internal loading and the implications of management alternatives. The further studies are outlined for modelling, field work, laboratory work, and pilot scale categories.

Pilot Scale Evaluation of Alum Injection

8. A pilot scale treatment of alum injection into sediments would be useful as it is needed to confirm the effect of chemical equilibrium and coagulation upon internal loading. It is an activity which would require substantial resources to measure the effectiveness of alum treatment in such an embayment as the Bay of Quinte. Monitoring of the effectiveness of treatment has only been done previously in whole lakes where mass balances can be used as the measurement tool. Such monitoring has also generally been incomplete.

6.10   Summary and Future Strategy

This study has shown that alum injection into sediments is effective in reducing internal phosphorus loading, based upon modelling and laboratory-scale experimentation.

There are, however, a number of factors and data deficiencies identified by this work which should be addressed before alum injection is considered for whole Bay addition. These data decificiencies have been identified above in section 6.9. Other aspects which would further evaluate include:

i)   the size of the bay relative to magnitude of the addition required (e.g., injection to selected portions of the upper bay, to the whole of the upper bay, or to the upper and middle bay);

ii)   potential environmental detriment (e.g., toxicity potential); and

iii)   the potential effectiveness of the null option under other RAP strategies.

A strategy of developing a critical path approach, carrying out key studies outlined in Section 6.9, and continual integration of these study results with other ecosystem oriented-studies and implementation of the Remedial Action Plan may be a most appropriate approach to solving problems resulting from internal loading on the Bay of Quinte. Improved controls of land based P under the Remedial Action Plan and carrying out key studies are viable options for remediating nutrient water quality problems in the short term. Over the long term, the null option (response of the Bay to further land-based control) and results of key studies, particularly pilot scale evaluation of alum addition, will establish the feasibility of alum injection for the long term.

7.0   CONCLUDING REMARKS

In this study, options for controlling internal loading have been considered. The options considered are:

o   the Null Option (Natural Clean-Up)
o   Chemical - Aluminum Salts
                   - Ferric Salts
                   - Calcium Salts
o   Biochemical Control
o   Fly Ash
o   Barriers
o   Dredging
o   Sediment Burial
o   Aeration
o   Lake Drawdown

These options are compared in Tables 7.1 to 7.3 with respect to their relative ease of implementation (Table 7.1), the cost of implementation (Table 7.2), and detrimental effects upon the social and natural environment. In this report, the effectiveness of different options was not evaluated for all options. However, a few comments upon the potential effectiveness of each option are illustrated in Table 7.4.

For purposes of the RAP, it might be useful to evaluate the potential effectiveness all of the control options. This could be considered in future work. In this study, 2 potentially cost effective systems were selected for evaluation of their potential effectiveness.

In the field of lake rehabilitation itself, dredging has been the typical technique used to control internal loading although most of the others (chemical, biochemical, fly ash, aeration, lake drawdown, and flushing) have also been used. At the moment, most new rehabilitation effects are directed towards aluminum injection. By way of comparison, Peterson (1981) concluded that aluminum injection was superior in overall economy and effectiveness to dredging, but that sediment removal (dredging) has the benefit of removing the nutrient source from the lake. Cooke et al. (1986) note that Peterson's conclusion of a high benefit-cost ratio for alum is supported by the effectiveness of alum treatment for a 10- to 12-year period in two lakes.

TABLE 7.1:   **RELATIVE EASE OF IMPLEMENTATION**

Option	Implementation	Relative Ease of Implementation
1 Null Option	No logistics	1
2 Chemical A Alum	Application every 3 years by a flotilla of small boats in liquid form	5
B Ferric		
C Calcium	Application yearly by a similar flotilla	
3 Biochemical	Application annually from a flotilla of small boata in granular form	5
4 Fly Ash	Application with an unknown frequency. It requires substantial quantities of material	8
5 Barriers	Placement and Removal of Barriers every year	7
6 Dredging	Removal of sediments with a smaller number of barges (relative to alum application)	2
7 Burial	Similar to dredging	2
8 Aeration	Placement and maintenance of aeration equipment. Less labour intensive than most other methods, but circulation is quite power intensive	4
9 Lake Drawdown	Cannot be implemented	-

TABLE 7.2:   **COST OF IMPLEMENTATION**

Option	Annual Operating Cost	Frequency of Operation	Comments
Null Option	$\emptyset$	$\emptyset$	No direct costs
Chemical (A) Alum	8 million	1/3 years	Major difference of alum and ferric are higher per unit costs
(B) Ferric	5.6 million	1/3 years	for ferric but higher dosage rates for alum
(C) Calcium	7 million	1/year	Substantially cheaper than Al or Fe.
Biochemical	100 million	1/year	Frequent application
Fly Ash	107 million	1/year	Appropriate application rate needs to be calculated based upon
	25 million	1/10 years	sediment modelling and results of point/diffuse source content
Barriers	2.2 million	1/year	Placed and removed each year. Assumes application to only one-half of the Bay. Effect on internal loading is not considered.
Dredging	28 million	1/10 years	Frequency of dredging needs to be more fully evaluated.
Burial	4 million	1/10 years	Detailed analysis; 50% safety factor gives $6 million/year.
	12 million	1/10 years	Based on typical dredging costs for Hamilton Harbour.
Aeration (a) Enhanced Circulation	75 million/yr	every year	Cost differences are caused by number of installations assumed; enhanced circulation assumes 272 compressors.
(b) Oxygenation	1.1 million/yr	every year	Oxygenation assumes 10 direct injection units.
Lake Drawdown	Not costed	-	Infeasible.

*Capital cost components amortized at rate of 12% per annum.

TABLE 7.3:   **COMPARISON OF DETRIMENTAL EFFECTS**

Control Option	Detrimental Effects
1 Null Option	o Continued loading maintains stressed ecosystem.
2 Chemical A Alum	o Generally negligible except for potential toxicity of aluminum
B Ferric	o Negligible
C Calcium	o Negligible
3 Biochemical	o Nitrate addition may affect ecosystem
4 Fly Ash	o Smothers existing benthic community o Heavy metal content is potentially toxic o Turbidity during placement o Altered sediment transport patterns of fine-grained fly ash
5 Barriers	o Potential impact upon boating o Socio concerns o Movement of bottom anchors will disturb sediments
6 Dredging	o Disturbs benthic community and fish habitat. o Potential impact of suspended solids and release of trace metals and toxic organics; field data show that these concerns can be minimized with well designed operations and disposal areas.
7 Sediment Burial	o Effects similar to fly ash (except no heavy metal concerns) and dredging
8 Aeration	o Minimal effect, except it has the potential to disturb sediments if not properly installed
9 Lake Drawdown	o Not evaluated

TABLE 7.4:   ILLUSTRATION OF THE EFFECTIVENESS OF
DIFFERENT CONTROL OPTIONS

Control Option	Points of Concern
1. Null Option	o Time is required for sediments to respond o Sediments will continue to give a long-term flux to water column.
2. Chemical	o Fe^{3+} is theoretically not effective under reduced conditions; however, field evidence indicates that it is partially effective.
3. Biochemical	o Increases aerobic/anoxic barrier to phosphorus diffusion. May not be too effective since upper Bay is relatively well oxygenated.
4. Fly Ash	o Extensive field/laboratory evidence is available from U.S. testing programs. o Neither chemical nor fly ash may be excessively effective if most of internal loading results from decomposition of freshly settled section in a micro-layer which accumulates unless fly ash application is each year.
5. Barriers	o Effectiveness depends upon minimizing the velocity of currents and waves and the role that erosion has in the magnitude of the internal loading.
6. Dredging	o Expensive for the whole lake. May be particularly useful if targeted to areas of high concentration (e.g., around STP discharges). Data describing areas of high concentration is not available.
7. Sediment Burial	o Same point as for Dredging.
8. Aeration	o May not be too effective since Upper Bay of Quinte is fairly well oxygenated.
9. Lake Drawdown	o Ineffective due to no possibility for water level manipulation.

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